July 2, 2004

Mr. Dustin Armstrong, Project Officer
Environmental Cleanup Program
Pennsylvania Department of Environmental Protection
Southeastern Regional Office
2 East Main Street
Norristown, PA 19401

Re: Final Supplemental Groundwater Characterization Report
       Bishop Tube Site
       East Whiteland Township, Chester County, Pennsylvania
       PADEP Contract No. ME359184
       Work Requisition No. 31-116

Dear Mr. Armstrong:

Baker Environmental, Inc. (Baker) is pleased to provide the Pennsylvania Department of Environmental Protection (PADEP) with the final version of the Supplemental Groundwater Characterization Letter Report. The report outlines the field procedures and analytical results for the additional investigative activities performed to evaluate the concentrations of chlorinated solvents contained in the bedrock aquifer underlying the Bishop Tube site. Recommendations regarding options for remediating the chlorinated solvents contained in the groundwater underlying the site are also presented for consideration.

1.0   BACKGROUND

1.1   Site Location and Setting

The Bishop Tube site is located along the east side of Malin Road approximately ¼ of a mile south of U.S. Route 30, in Frazer, East Whiteland Township, Chester County, Pennsylvania. The site can be located on the Malvern, Pennsylvania USGS 7.5-Minute Quadrangle Topographic Map at north 40^0 02' 24" latitude and west 75^0 32' 13" longitude (see Figure 1). The Central and Western Chester County Development Authority (CWCCDA) currently owns the site. The CWCCDA acquired the property from Christiana Metals in late 2002.

Survey mapping indicates that the current property is approximately 13.7 acres in size. The Bishop Tube site is situated in a suburban area that is mainly served by public water. Some local residents and businesses, however, still rely upon private wells for their water supply needs. According to the United States 2000 Census report for Chester County Pennsylvania, 9,333 people were listed as residing within the East Whiteland Township, Pennsylvania area (United States Census Bureau, 2000).

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The Bishop Tube site is situated within a southwest-northeast trending valley locally referred to as the Chester Valley area. This valley is mainly underlain by easily eroded rocks comprised of limestone and dolomite. The northwestern edge of Chester Valley is flanked by resistant quartzites that form the North Valley Hills. The southeastern edge of the valley is bordered by a combination of resistant phyllites and schists, and is locally referred to as the South Valley Hills. The main trunk streams draining the valley are Little Valley Creek and Valley Creek. The headwaters of Little Valley Creek originate along the upper portion of the hillside immediately east of the Bishop Tube site.

The Bishop Tube facility was formerly used to process precious metals and to fabricate stainless steel specialty items, namely tubing and piping products. The site includes two large out-of-service rectangular-shaped one-story concrete block buildings that cover approximately 3.2 acres of surface area. These two buildings are connected to one another and are referred to as Building #5 and Building #8. A considerable amount of cutting and filling has occurred at the site to construct the buildings and parking areas. Building #5 was constructed in 1950 and Building #8 was built in 1959. The remainder of the property primarily consists of paved and gravel-covered storage/parking areas, with a smaller amount of undeveloped grassy areas. An 8-foot high chain-link fence currently borders the northern, southern, and western edges of the property (see Figure 2).

A transformer pad is located next to a loading dock at the southern side of Building #5. The former boiler room for Building #5 is located just east of the transformer pad and loading dock. Two underground storage tanks (UST) used to store No. 2 and No. 6 fuel oils are believed to exist along the western and southern sides of this boiler room. Each of these two USTs is reportedly 5,000 gallons in capacity. A third UST (20,000-gallon capacity) used to store fuel oil for the boiler room of Building #8 is believed to exist in the middle of the plant complex between Buildings #5 and #8.

During the former plant operations, stainless steel was first cleaned prior to fabrication by passing the raw materials through several pickle tanks (i.e., acid). The former pickle tank area is located in the eastern portion of Building #8. An 8-inch raised concrete pad currently overlies the area where the pickling operations were conducted. Rinse waters from the pickling process were reportedly mixed/aerated with the sanitary wastes generated by the plant facility and discharged to an underground sanitary cesspool located between the east end of Building #5 and the concrete "acid" aboveground storage tank (AST) pad.

Immediately east of Building #5 is a concrete-covered area formally used to store drums of solvents and chemicals associated with the plant operations. A raised concrete berm surrounds the eastern portion of the former drum storage area. Several rectangular-shaped concrete pads exist within the bermed enclosure. According to plant records, several aboveground storage tanks were housed in this area for the storage of nitric acid (4,000-gallon capacity), hydrofluoric acid (5,100-gallon capacity), used acids (4,000-gallon capacity), and acid rinse waters (two tanks, each 5,600-gallon capacity). These acids and waste fluids were apparently associated with the former pickling operations performed at the site.

Along the northern edge of Building #8 are two 4 feet by 4 feet concrete-covered areas. According to plant records, a 4,000-gallon capacity aboveground storage tank rested on support pillars in this area for the storage of trichloroethylene (TCE). TCE was transferred from the aboveground storage tank to the vapor degreaser located within Building #8 via a 1¼-inch carbon steel underground pipe.

At the west end of Building #8 is a cooling tower. During the time period from June 2001 through December 2003, groundwater from several springs was observed emerging through cracks in the asphalt

pavement of the parking area situated east of the cooling tower. Hydrophytic vegetation (i.e., cattails) is well established and reddish-brown iron staining is present on the surface of the asphalt pavement adjacent to the points where groundwater emerges from the subsurface and along the downslope runoff areas. Thin films (i.e., sheens) were observed floating on top of the water discharging from these springs. In addition, odors resembling weathered hydrocarbon compounds have been observed on a frequent basis in this area. It should be noted that the Mobil Oil Corporation, Inc. currently operates a bulk oil terminal on the property situated immediately west of Malin Road adjacent to the Bishop Tube site. A review of aerial photographs (Baker, 2002a, and Baker, 2002b) suggests that the petroleum products have been stored in aboveground storage tanks at this facility since at least 1947. Information furnished by the PADEP indicates that hydrocarbon compounds have been released into the environment at the Mobil bulk oil terminal. A quarterly groundwater sampling report, dated January 2000, prepared by Handex, Inc. on behalf of Mobil shows that the groundwater underlying the Mobil bulk oil terminal flows to the north-northeast in the direction of the Bishop Tube site. The analytical results for the groundwater samples collected from the monitoring wells at the Mobil bulk oil terminal during the period from October through December 1999 show that the groundwater locally contains concentrations of hydrocarbon compounds (i.e., benzene, toluene, ethylbenzene, methyl tertiary butyl ether, cumene, and naphthalene) exceeding the PADEP Non-Residential Statewide Health-based Standards. Separate-phase liquid hydrocarbons were reported by Handex, Inc. to be floating on top of the groundwater contained in several monitoring wells at the Mobil bulk oil terminal. In December 1999, removal of liquid hydrocarbon compounds via pneumatically operated skimmer pumps was actively being performed at the Mobil bulk oil terminal from monitoring wells MW-1, MW-2, and MW-3 (Handex, 2000).

The northern and southern edges of the Bishop Tube property are bordered by railroad tracks maintained by Norfolk Southern and Amtrak, respectively. Malin Road borders the western edge of the site. A bulk fuel oil terminal, operated by the Mobil Oil Corporation, is situated along the western side of Malin Road next to the Bishop Tube site (see Figures 1 and 2).

Topography decreases from a high of approximately 500 feet above mean sea level near the Amtrak railroad tracks at the southern boundary of the Bishop Tube site to a low of 370 feet above mean sea level along Little Valley Creek situated at the northeast corner of the property. Based upon these topographical differences, surface water runoff is in a north-northeasterly direction across the site. Little Valley Creek receives surface water runoff from the parking areas situated along the east side of the manufacturing building. A drainage channel is present immediately north of Building #8 adjacent to the Norfolk Southern railroad tracks. This drainage channel receives runoff from the rooftop and parking areas surrounding Building #8, and ultimately conveys this surface water to Little Valley Creek situated along the eastern edge of the property.

1.2   Site History

Prior to the construction of the plant buildings, land use of the Bishop Tube property was primarily agricultural in nature. Manufacturing operations began at the site under the name of the "J. Bishop and Company, Platinum Works" in 1951. Little is known about the early manufacturing work performed at the site. Industrial operations are believed to have included the processing of platinum and other precious metals.

In 1967, the plant was sold to Matthey Bishop and Company. At this time, the industrial operations performed at the site were changed to encompass the manufacturing of special seamless stainless steel

tubing. Under these new operations, the plant was classified as a redraw mill, where stainless steel pipe was reduced to specific diameters and wall gauges by successive redraws and heat treatment. Matthey Bishop and Company sold the plant in 1969 to the Whittaker Corporation.

In 1974, the Christiana Metals Corporation purchased the manufacturing plant. Christiana Metals continued to operate the stainless steel tube manufacturing business at the site until the early 1990's, when the building and facilities were sold to the Marcegaglia Group, USA-Damascus Division. The plant operated under the name of the Damascus-Bishop Tube Company, Inc. from early 1990's to the closure of the business in 1999. The site is currently owned by the Central and Western Chester County Development Authority and is non-operational.

Manufacturing operations performed at the Bishop Tube facility included the cleaning, pointing, shaping (i.e., drawing), welding, degreasing, annealing, straightening, sandblasting, polishing, and painting of stainless steel and specialty metals into tubes (i.e., pipes) and other various metal products. The plant reportedly used a wide variety of materials, including nitric acid, hydrofluoric acid, caustic materials (water treatment), motor oil (20W40), gear oils, specialty drawing lubricants, degreasing solvents (TCE), anhydrous ammonia, coolants, polishing compounds, metal alloys, and paints. The processing procedures included a cleaning/pickling operation to prepare metals for fabrication (source of acidic and caustic solutions/sludges containing heavy metals); drawing operations (introduction of lubricants); trimming operations (source of heavy metal cuttings and dusts); a degreasing/cleaning operation (source of chlorinated solvents and waste oils); an annealing/heat treatment and subsequent cooling operation (source of heavy metal scales); a trim, ream, and deburr operation (source of heavy metal cuttings and dusts); and finally a polishing operation for finished metals (source of metal dusts).

An engineering report dated April 9, 1973, identified the following industrial wastewater sources generated by the manufacturing operations at the Bishop Tube facility:

- Furnace and degreaser cooling waters.
- "A"-furnace cooling water.
- Compressor and after cooler cooling water.
- Pickle rinse water and floor drains.
- Boiler blow down.

Prior to the installation of a public sewer system in the area, plant sanitary wastes were apparently directed to an aeration tower and pit where the sewage was circulated for treatment. A waste acceptance firm pumped out the sludges that settled to the bottom of the pit approximately once every six months. The treated liquid effluent flowed by gravity to a holding tank where it was then pumped to an underground cesspool for disposal. This underground cesspool is believed to be in the area between the east end of Building #5 and the concrete "acid" aboveground storage tank (AST) pad.

The cooling water waste streams (i.e., furnace and degreaser cooling waters, "A"-furnace cooling water, and compressor and after cooler cooling water) were combined into two discharges to Little Valley Creek. The furnace and degreaser cooling water was apparently directed from the manufacturing complex to Little Valley Creek via a 12-inch diameter asphalt-coated corrugated metal pipe. The non-contact cooling water (approximately 60 gallons per minute) from the heat exchangers was discharged to Little Valley Creek under a National Pollutant Discharge Elimination System (NPDES) Permit (No. PA0013641). The

"A"-furnace cooling water and compressor and after cooler cooling water were combined and discharged to the drainage swale along the Norfolk Southern Railway line north of Building #8.

Liquid overflow wastes originating from the pickle acid-rinse tanks and floor drains were discharged into a holding tank where they were reportedly mixed with the sanitary effluent from the aeration tower. The mixed liquid wastes contained in the holding tank were reportedly pumped to the underground cesspool located at the east end of Building #5 for disposal. Samples of the pickle acid rinse stream collected in 1973 showed that this liquid waste was acidic (pH of 2.0), and contained high concentrations of nitrates and fluorides, reflecting the characteristics of the nitric-hydrofluoric acid pickle liquor. In addition, the pickle acid rinse stream also contained elevated concentrations of iron (89.9 mg/l), nickel (12.5 mg/l), and total chromium (17.2 mg/l).

Blow down of the two low-pressure boilers was reportedly conducted once or twice a day, depending upon the concentration of the dissolved solids. The boiler blow down discharges were directed to a small pit/impoundment under the driveway. This water apparently then overflowed onto the driveway and evaporated. The location of this pit/impoundment is believed to have been located under the asphalt driveway east of Building #8. Field notes, recorded by an inspector from the Pennsylvania Department of Environmental Resources (PADER – now the PADEP), suggest that this underground pit/impoundment may have also received overflow discharges from the sanitary cesspool, pickle area floor drains, and furnace and degreaser cooling waters. Fluids contained in the underground pit/impoundment may have been pumped to a second cesspool area located within the alcove east of the boiler room in Building #5. Overflow from this second cesspool was conveyed by a stormwater drainpipe to Little Valley Creek east of the site. Bishop Tube closed the pit/impoundment and cesspool areas in 1979. Closure reportedly involved filling these impoundments with limestone and covering the waste disposal areas with concrete.

1.3   Previous Investigations

Previous environmental investigations conducted at the site on behalf of the former property owners have identified impacts to soils and groundwater related to the past manufacturing operations. Specifically, elevated concentrations of chlorinated solvents (i.e., trichloroethylene, 1,1,1-trichloroethane, tetrachloroethylene) and fluoride have been detected in the soils and groundwater at the site that exceed the PADEP Statewide Health-based Standards. In addition, surface water and sediment samples collected from Little Valley Creek have also been found to contain elevated concentrations of chlorinated solvents and fluoride that exceed the established regulatory standards.

In 1972, elevated concentrations of fluoride were detected in surface water samples collected from Little Valley Creek by the PADER. These samples were collected as part of a regional study to evaluate the quality of water within the Little Valley Creek drainage basin. The source for the fluoride was traced to a 12-inch diameter discharge pipe located just east of the Bishop Tube site. Additional water quality studies performed in the early 1980's confirmed the presence of elevated concentrations of fluoride in Little Valley Creek. To evaluate the groundwater conditions at the Bishop Tube site, Betz, Converse, and Murdoch (BCM), Inc., on behalf of Christiana Metals, drilled four monitoring wells (MW01 through MW04) at the site. The results of this investigation determined that the groundwater in the vicinity of MW04 (drilled adjacent to the pit/impoundment east of Building #8) contained an elevated concentration (23.1 mg/l) of fluoride. This finding suggested that the discharge of groundwater via baseflow from the fractured bedrock aquifer underlying the Bishop Tube site might be the source of the fluoride in Little

Valley Creek. To address this concern, BCM, Inc. recommended that groundwater samples be collected from the monitoring wells on a periodic basis to monitor the concentrations of fluoride.

On June 10, 1981, nitric and hydrofluoric acid were inadvertently mixed causing a release of an acid mist at the Bishop Tube facility. The acid mist drifted offsite and resulted in the evacuation of approximately 500 residents of the nearby General Warren Village housing development (see Figure 2). Sixteen people were treated at a local hospital for respiratory problems associated with the release of the acid vapors.

In 1984, fluoride was detected in groundwater that had infiltrated through cracks into a concrete-lined sump within the manufacturing plant. Due to an inadvertent connection between the sump and the effluent pipe for non-contact cooling water, fluoride was discharged to Little Valley Creek. The concentration of fluoride dissolved in the effluent water exceeded the permitted average monthly limit of 10 mg/l, and was in violation of Bishop Tube's NPDES permit. To correct the situation, the sump water was transferred to a temporary storage tank for treatment/disposal at an offsite permitted facility.

In 1987, BCM/Smith, Inc., on behalf of Christiana Metals Corporation, conducted an investigation to update the 1981 groundwater study, and to evaluate potential impacts from degreasing agents (i.e., chlorinated solvents) historically used at the plant. The investigation included the drilling of five additional groundwater monitoring wells (MW05 through MW09), the collection of groundwater samples from the monitoring well network, the drilling of five borings and collection of soil samples, and the collection of three surface water samples from Little Valley Creek. The results of this investigation indicated the presence of volatile organic compounds (VOCs), primarily trichloroethylene (TCE) and 1,1,1-trichloroethane (1,1,1-TCA) in the groundwater underlying the site. Heavy metals were also detected in the soil and groundwater samples collected during the investigation (BCM/Smith, 1988).

On July 26, 1988, a meeting was conducted between the PADER, BCM/Smith, Inc., and Christiana Metals Corporation to discuss the environmental concerns at the Bishop Tube site. At this meeting BCM/Smith presented the PADER with a work plan for performing further investigative work to delineate the extent of VOCs (i.e., TCE and 1,1,1-TCA) dissolved in the groundwater underlying the site. The PADER requested that quarterly groundwater monitoring also be performed in the vicinity of the east end of the manufacturing plant where cooling water was being discharged to Little Valley Creek. This monitoring was apparently requested to document the concentrations of fluoride in groundwater that were formerly predicted to decrease with time as a result of improvements made in the plant's pickle liquor handling practices. In October 1988, prior to the submission of a revised work plan, a soil vapor survey (SVS) was conducted along the north side of the manufacturing building. The results of the soil vapor survey showed low concentrations of TCE, tetrachloroethylene (PCE), and trans-1,2-dichloroethylene (trans-1,2-DCE) in the soils underlying the former TCE aboveground storage tank (AST), loading dock area, and concrete storage pad. A final groundwater remediation work plan was submitted to the PADER in June 1989.

In 1989, BCM/Smith, on behalf of Christiana Metals Corporation, drilled and installed seven additional monitoring wells (MW10 through MW16) at the Bishop Tube site. This work included five shallow monitoring wells and two deep monitoring wells. To evaluate the concentrations of VOCs contained in the soils where the SVS was performed, BCM/Smith drilled eight borings at the site. Five of these soil borings were drilled along the north side of Building #8, and three soil borings were drilled inside Building #8 in the vicinity of the vapor degreaser. The soil samples collected from the borings drilled in

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the vicinity of the vapor degreaser showed that the subsurface materials contained elevated concentrations (up to 3,280 mg/kg) of TCE (O'Brien and Gere, 1998).

Groundwater samples were collected from the monitoring wells at the Bishop Tube facility in 1989. The sampling results were presented to the Christiana Metals Corporation in a report prepared by BCM/Smith titled: *Results of Implementation of Groundwater Remediation Work Plan, Phase I*, dated January 1990. This report characterized the water quality conditions in the underlying bedrock aquifer and the direction of groundwater flow at the site. The sampling results indicated that groundwater in the underlying bedrock aquifer contained elevated concentrations of chlorinated solvents (namely TCE). Moreover, the indoor vapor degreaser area was suspected as the principle source for the VOCs contained in the groundwater, with the outdoor former TCE AST being a potential secondary source.

In 1995, BCM/Smith on behalf of the Christiana Metals Corporation drilled four additional monitoring wells (i.e., MW17, MW18, MW19, and MW20) to evaluate the water quality conditions at the Bishop Tube facility. These wells included two lower (i.e., deep) bedrock monitoring wells (i.e., MW17 offsite and MW19 onsite), one onsite upper bedrock monitoring well (MW18), and one onsite shallow overburden well (MW20). During the drilling of the borehole for MW19, continuous rock cores were collected from the interval between 300 feet to 500 feet below grade. The investigation also included downhole video surveys in monitoring wells MW17 and MW19 (i.e., deep wells), packer testing and the collection of depth discrete groundwater samples in monitoring wells MW17 and MW19, slug testing within the packer test interval in monitoring wells MW17 and MW19, and a 30 hour aquifer test in monitoring well MW19. The results of the investigation indicated that high yielding fracture zones occur within the upper portion of the bedrock aquifer within 100 feet of the ground surface. The concentrations of TCE detected in the groundwater samples collected from deeper intervals of MW17 and MW19 were similar to the shallow groundwater samples. The groundwater samples collected from monitoring wells MW17 and MW19 were found to contain concentrations of TCE ranging between 44 mg/l and 680 mg/l. These concentrations are equivalent to 4% to 62% of the pure-phase aqueous solubility limit of TCE. This information suggests that dense non-aqueous phase liquids (DNAPLs) may be present within the upper portion of the bedrock aquifer underlying the site (O'Brien and Gere, 1998).

In 1996, Smith Environmental (formerly BCM) collected surface water samples from Little Valley Creek in the vicinity of the Bishop Tube site. The sampling data for these surface water samples were presented in the 1998 *Site Characterization and Interim Remedial Action Plan* report, prepared by O'Brien and Gere, Inc. following the bankruptcy of Smith Environmental. These surface water samples were analyzed for fluoride and VOCs. The analytical results showed that TCE was detected in the surface water of Little Valley Creek at concentrations of 0.075 mg/l and 0.01 mg/l. These concentrations exceed the Surface Water Quality, Human Health-based Standard for TCE of 0.003 mg/l.

In January of 1996, Smith Environmental collected a groundwater sample from a domestic water supply well, CH1985, located at 54 Conestoga Road. This well is reportedly 320 feet deep with 36 feet of casing. The groundwater samples collected from this well were found to contain concentrations of the following VOCs: TCE at 0.053 mg/l, 1,1,1-TCA at 0.0081 mg/l, and 1,1-DCA at 0.0011 mg/l. The sampling data show that the concentration of TCE exceeded the Act 2 Statewide Health-based Groundwater Standard of 0.005 mg/l. In 1999, a whole-house carbon filtration system was installed in this home at the expense of the Christiana Metals Corporation.

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In 1998, O'Brien and Gere, on the behalf of the Christiana Metals Corporation, completed a *Site Characterization and Interim Remedial Action Plan* report outlining a planned interim remedial action program to address the groundwater quality conditions at the Bishop Tube site. This report was prepared using the information compiled from previous investigations performed by other consultants at the site. The interim remedial action program proposed extracting groundwater from the north and east plant areas in an attempt to limit the offsite migration of the TCE plume. The final number and locations of the extraction wells were to be based upon a pilot testing program and the performance data of the initial phase-in implementation program. The conceptual interim remediation plan included the extraction of groundwater from two vertically oriented pumping wells. One extraction well would be located in the vicinity of well cluster MW02/MW03/MW19 and pumped at a rate of 30 to 60 gallon per minute (gpm). A second extraction well would be located along the eastern side of Building #8 and pumped at a rate of 30 gpm. The plan also included the possibility of using a horizontally oriented well to extract groundwater within the upper portion of the overburden/shallow bedrock. The groundwater extracted from the pumping wells was proposed to be treated onsite using an air stripper. The treated effluent would ultimately be discharged to the local publicly owned treatment works (POTW). A catalytic oxidizer with a caustic scrubber was proposed to reduce the concentrations of VOCs in the off-gas generated by the air stripper. This interim remedial action plan was never implemented.

During the period of June through October 2001, a Site Characterization was performed by Baker on behalf of the PADEP to determine the concentrations of organic and inorganic compounds contained in the soils, sediments, surface water, and shallow groundwater at the Bishop Tube site (Baker, 2002a). The results of this investigation were used to evaluate potential risks to human health and the environment. A total of 87 soil borings were drilled inside and around the perimeter of the plant building to evaluate the horizontal and vertical extent of organic and inorganic compounds contained in the soils. To assess potential impacts resulting from the past disposal of waste materials at the site, eight sets of surface water and sediment samples were collected from selected locations along Little Valley Creek as well as the drainage swale situated north of Building #8. Elevated concentrations of chlorinated solvents (trichloroethylene, tetrachloroethylene, 1,1-dichloroethene, 1,2-dichloroethene, 1,1,1-trichloroethane, 1,1,2-trichloroethane, 1,2-dichloropropane, and vinyl chloride); hydrocarbon compounds (MTBE, benzene, and toluene); polynuclear aromatic hydrocarbon compounds (anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, and pyrene); polychlorinated biphenyls (aroclor 1260); and heavy metals (antimony, arsenic, barium, beryllium, cadmium, chromium, copper, lead, mercury, nickel, silver, thallium, vanadium, and zinc) exceeding the PADEP Health-based Standards were detected in the soils, sediments, surface water, and shallow groundwater at the site. The results of the Phase I Site Characterization were used by the PADEP to develop a Scope of Work for further investigating the impacts to the groundwater underlying the Bishop Tube site.

During the period from December 2001 through June 2002, Baker performed a Groundwater Investigation at the Bishop Tube site on behalf of the PADEP (Baker, 2002b). This investigation included the following work: 1) the collection of groundwater samples from selected private water supply wells and springs believed to be situated hydraulically downgradient to the direction of groundwater flow from the Bishop Tube site; 2) the performance of geophysical well logging techniques to log the boreholes of monitoring wells MW09, MW17, and MW19; 3) the completion of a seismic refraction geophysical survey to map the elevation of the top of bedrock for selected areas at the site; 4) the installation of four new monitoring wells (MW21, MW22, MW23, and MW24); 5) the collection of one round of groundwater samples from the monitoring well network to assess the concentrations of organic and inorganic compounds contained in the underlying fractured bedrock aquifer; and 6) the performance of a

24-hour constant rate pumping test to evaluate the hydraulic properties of the shallow groundwater flow system.

The results of the seismic refraction geophysical survey showed that the surface of the bedrock underlying the former vapor degreaser area in Building #8 and the former drum storage area is pinnacled. The elevation changes exhibited by the surface of the bedrock were postulated to be providing a path for the migration of DNAPLs in the subsurface. A trough/depression was identified in the southeast corner of Building #8 by the geophysical survey. This trough/depression was suspected to represent the location of the former waste disposal lagoon.

The results for the geophysical logging responses recorded abrupt increases in the specific conductance of the groundwater with respect to depth in the boreholes for former deep monitoring wells MW17 and MW19. This information suggested that separate-phase DNAPLs may be present in the deeper portions of the fractured bedrock aquifer underlying the site.

The data collected during the 24-hour constant rate aquifer test showed that the average hydraulic conductivity of the weathered bedrock/saprolite interval is approximately 2.06×10^{-2} ft/min. The average hydraulic conductivity value estimated for the fractured bedrock interval is 4.19×10^{-3} ft/min. In addition, the aquifer testing results showed that groundwater flow within both the saprolite/weathered bedrock and the fractured bedrock intervals underlying the Bishop Tube site is anisotropic. Specifically, the hydraulic conductivity within these intervals was found to be 1.5 times greater parallel to the strike of the rock beds than perpendicular to strike.

The analytical results for the groundwater samples collected from the offsite residential wells and springs showed that elevated concentrations of TCE (37 μg/l) and PCE (5.8 μg/l) exceeding the PADEP Statewide Health-based Groundwater Standards were detected in well CH1985 (54 Conestoga Road) and the spring at sample point No. SP-49 (10 Winding Way), respectively. The source of the TCE and PCE detected in well CH1985 and the spring at sample point No. SP-49 is unknown. Based upon published groundwater flow maps for the Chester Valley, Pennsylvania area, well CH1985 is believed to be situated hydraulically downgradient to the direction of groundwater flow from the Bishop Tube site. The spring at sample point SP-49 is situated lateral to the direction of groundwater flow from the Bishop Tube site.

The groundwater samples collected from the monitoring wells in February 2002 were found to contain concentrations of the following VOCs that exceeded the PADEP Non-Residential Statewide Health-based Groundwater Standards: 1,1-DCE, 1,2-DCE, methylene chloride, PCE, 1,1,1-TCA, TCE, and vinyl chloride. The concentrations of TCE were found to exceed the PADEP Non-Residential Statewide Health-based Groundwater Standard in each monitoring well at the Bishop Tube site with the exception of the samples collected from monitoring wells MW01 (upgradient) and MW24. The highest concentration of TCE (i.e., 45,000 μg/l) was detected in the groundwater sample collected from monitoring well MW22. At the time of the investigation, monitoring well MW22 was the deepest well at the Bishop Tube site. Because the concentrations of TCE detected in the groundwater samples collected from monitoring well MW22 are greater than 10% of the pure-phase solubility limit for TCE, separate-phase DNAPLs are suspected to be present in the fractured bedrock aquifer underlying this area of the site. The results of the Phase II Groundwater Investigation were used by the PADEP to develop a Scope of Work for further investigating the impacts to the soils and groundwater underlying the Bishop Tube site.

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A Supplemental Soil Characterization (Baker, 2003) was performed at the Bishop Tube site by Baker during the period between September 2002 and November 2002 to: 1) determine changes in the elevation of the bedrock surface in uncharacterized areas at the site (potential DNAPL accumulation points and/or migration conduits); 2) define the lateral limits of the VOCs contained in the subsurface materials underlying the former vapor degreaser area #2 in Building #5, the former vapor degreaser area #1 in Building #8, and in the former drum storage area; 3) characterize the concentrations of organic and inorganic compounds contained in the subsurface materials overlying the depression occurring within the top of bedrock situated at the southeast corner of Building #8 (suspected former waste disposal lagoon area); and 4) determine the presence or absence of separate-phase DNAPLs occurring within the soils and weathered bedrock materials underlying the site.

The supplemental investigative activities for further characterizing the soils included: performing additional geophysical survey studies in uncharacterized areas at the site; the drilling of additional soil borings using membrane interface probe (MIP) technology; the collection of confirmatory soil samples using direct push technology to verify the concentrations of VOCs contained in the subsurface materials; and the deployment of reactive non-aqueous phase liquids (NAPL) liners (manufactured by Flexible Liner Underground Technologies, Ltd. Company [FLUTe®]) in selected borings to determine the presence of separate-phase DNAPLs contained in the soils.

The Supplemental Soil Characterization provided the following information:

- The instrument responses recorded from the borings drilled using MIP technology show that VOCs extend from near surface sources to the top of bedrock. This information is consistent with the analytical results for the soil samples collected during the Phase I Site Characterization (Baker, 2002a) and suggests that DNAPLs have migrated downward and invaded the underlying fractured bedrock aquifer.

- The MIP testing data were used to select sampling locations along the perimeter of the former vapor degreaser area #2 in Building #5, the former vapor degreaser area #1 in Building #8, and in the former drum storage area. The analytical results for the soil samples collected along the edges of these three areas of concern were used to define the lateral extent of TCE contained within the soils.

- A total of seven FLUTe® NAPL liners were installed at selected locations at the site during the investigation. Evidence of staining suggesting the presence of DNAPLs was observed on the NAPL liners deployed in the DSA-FLUTE01 (i.e. drum storage area) and VDP-FLUTE02 (i.e., vapor degreaser #1 area) boreholes. This information confirms the presence of residual DNAPL fluids contained in the soils surrounding these drilling locations. Staining was not observed on the NAPL liners deployed in the following boreholes: AST-FLUTE01, SDA-FLUTE01, VDP-FLUTE01, VD2-FLUTE01, and DSA-FLUTE02.

- The results provided by the NAPL liner testing showed that perched separate-phase pools of DNAPLs probably do not exist on top of the bedrock surface underlying the former vapor degreaser area #1, the former vapor degreaser area #2, and the former drum storage area. This finding suggests that the fractures contained in the bedrock underlying each of these three

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areas has allowed the chlorinated solvents to migrate downward into the underlying fractured bedrock aquifer.

- Soil samples were collected from each of the three principal areas of concern (i.e., former vapor degreaser area in Building #5, the former vapor degreaser area in Building #8, and the former drum storage area) during the investigation to evaluate the physical characteristics of the subsurface materials. These testing data were used to evaluate the potential presence of separate-phase DNAPLs (using the principals of equilibrium partitioning) and the total mass of TCE contained in the soils underlying each of the three principal areas of concern at the Bishop Tube site. Applying the principals of equilibrium partitioning, calculations were performed to assess the potential presence of DNAPLs using the analytical data for the soil samples collected from the borings previously drilled at the site. The comparison of the maximum concentrations of TCE detected in the soils to the calculated C_T^{NAPL} values suggests that free-phase DNAPLs may be present in the saturated soil/weathered bedrock materials underlying the former vapor degreaser area #1 and the former drum storage area. The testing results for the FLUTe® NAPL liners showed that perched separate-phase pools of DNAPLs probably do not exist on top of the bedrock surface underlying these two locations. This information collectively suggests that the DNAPLs contained in the soils underlying the former vapor degreaser area #1 and the former drum storage area may exist as isolated residual NAPL particles/globules occurring within the pore spaces of the soil/weathered bedrock materials.

- An estimated volume of 113,265 cubic feet (ft^3) (i.e., 4,195 cubic yards [yds^3]) of soil materials containing 500 µg/kg or more of TCE underlie the former vapor degreaser area #1 in Building #8. Based upon a dry bulk density value of 111.1 pounds per cubic foot (lbs/ft^3), this volume of soil equates to a weight of 12,587,608 pounds (6,294 tons) of impacted soil materials. Approximately 4,505 pounds of TCE is estimated to occur in the soils underlying the former vapor degreaser area #1. An estimated volume of 12,240 ft^3 (i.e., 453 yds^3) of soil materials containing 500 µg/kg or more of TCE underlie the former vapor degreaser area #2 in Building #5. Based upon a dry bulk density value of 112.4 lbs/ft^3, this volume of soil equates to a weight of 1,375,566 pounds (688 tons) of impacted soil materials. Approximately 15 pounds of TCE is estimated to occur in the soils underlying the former vapor degreaser area #2. An estimated volume of 123,076 ft^3 (i.e., 4,558 yds^3) of soil materials containing 500 µg/kg or more of TCE underlie the former drum storage area. Based upon a dry bulk density value of 116.8 lbs/ft^3, this volume of soil equates to a weight of 14,369,515 pounds (7,185 tons) of impacted soil materials. Approximately 2,911 pounds of TCE is estimated to occur in the soils underlying the former drum storage area.

Based upon the results of the Supplemental Soil Characterization the following recommendations were provided to the PADEP:

1. The results of the environmental studies conducted previously at the site (Baker, 2002a and 2002b) including the findings of the Supplemental Soil Characterization show that the soils underlying the former vapor degreaser area #1 (Building #8), the former vapor degreaser area #2 (Building #5), and the former drum storage area contain elevated concentrations of chlorinated solvents (namely TCE). The concentrations of chlorinated solvents contained in

the soils underlying these areas of concern are believed to be functioning as residual sources for the TCE dissolved in the shallow portion of the fractured bedrock aquifer underlying the site. Based upon this finding, Baker recommended that the Department consider implementing a Feasibility Study to evaluate different remedial techniques to reduce the concentrations of chlorinated solvents contained in the soils.

2. The information provided by Supplemental Groundwater Characterization should be used in selecting an appropriate technology for the remediation of the soils at the site.

3. Importantly, the screening and selection of appropriate remedial technologies should consider what impact these techniques may have on the future site development plans.

1.4   Purpose and Objective

The purpose of the Supplemental Groundwater Characterization was to more fully assess the following issues: 1) evaluate the characteristics of the fractured bedrock aquifer underlying the site; 2) determine the presence or absence of separate-phase DNAPLs occurring within the fractured bedrock aquifer; 3) evaluate changes in the concentrations of chlorinated solvents with respect to depth within the fractured bedrock aquifer; 4) assess seasonal fluctuations in the elevation of groundwater and the concentrations of chlorinated solvents dissolved in the fractured bedrock aquifer; and 5) evaluate the characteristics of groundwater flow and the migration of chlorinated solvents dissolved in the fractured bedrock aquifer by developing a groundwater flow model. The additional information provided by the Supplemental Groundwater Characterization was evaluated with the environmental data collected during previous investigations to assess various remediation techniques for cleaning up the elevated concentrations of chlorinated solvents contained within the fractured bedrock aquifer underlying the area in the vicinity of the Bishop Tube site.

2.0   SITE ASSESSMENT

The scope of the field procedures performed during the Supplemental Groundwater Characterization is outlined in the following sections.

2.1   Site Scoping Meeting and Historical Information Review

Baker received a letter from the PADEP on June 13, 2002 to prepare a Work Plan and Cost Proposal (i.e., Change Order #9) for performing supplemental investigation activities associated with further characterizing the soils and groundwater at the Bishop Tube site. The supplemental investigative activities requested by the PADEP for further characterizing the soils and groundwater included: 1) performing additional geophysical survey studies in uncharacterized areas at the site; 2) drilling of additional soil borings using membrane interface probe (MIP) and direct-push technologies; 3) collecting of confirmatory soil samples to characterize the lateral extent of chlorinated solvents contained in the soils; 4) deploying Flexible Liner Underground Technologies, Ltd. Company (FLUTe®) non-aqueous phase liquids (NAPL) liners in selected borings to determine the presence of separate-phase DNAPLs contained in the soils; 5) drilling and installation of four multi-screened monitoring wells to more fully evaluate changes in water quality with respect to depth within the underlying fractured bedrock aquifer; 6) the deployment of Flexible Liner Underground Technologies, Ltd. Company (FLUTe®) non-aqueous phase liquids (NAPL) liners in the open borehole sections two monitoring wells to determine the

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presence/absence and the depth of occurrence of potential DNAPLs in the fractured bedrock aquifer; 7) the collection of groundwater samples on a quarterly basis to evaluate the concentrations of TCE within the fractured bedrock aquifer underlying the site; and 8) the development of a three-dimensional groundwater flow model for the area of Chester Valley surrounding the Bishop Tube site to evaluate the regional direction(s) of groundwater flow and to estimate the rate of migration of TCE within the fractured bedrock aquifer. It should be noted that Baker completed the investigation of the soils underlying the Bishop Tube site under a separate phase of the work proposed under Change Order #9 (Baker, 2003). The Work Plan and Cost Proposal for Change Order #9 were prepared as part of the project planning task of the Work Order. In addition, Baker also prepared an addendum to the original site-specific Health and Safety Plan (HASP) dated June 5, 2001.

Copies of the draft version of the Work Plan and Cost Proposal for Change Order #9 were submitted to the PADEP Southeastern Regional Office for review on July 24, 2002. The final versions of the Work Plan and Cost Proposal for Change Order #9 were submitted to the PADEP on August 23, 2002. The PADEP issued authorization (i.e., "Notice to Proceed") for Baker to begin the Supplemental Site Characterization activities included in Change Order #9 on September 18, 2002.

Pursuant to Change Order #9, the fieldwork activities associated with the Supplemental Soil Characterization at the Bishop Tube site began in September 2002. Drilling and project-related work associated with the Supplemental Groundwater Characterization began in December 2002 and is scheduled to be completed in late February 2004.

2.2   Field Procedures

The various field procedures, operations, and methods used by Baker to complete the project task objectives outlined in the Scope of Work for Change Order #9 are presented in the following sections.

2.2.1   Introduction

An intrusive field investigation was performed to define the vertical extent and to further characterize the concentrations of the VOCs dissolved in the fractured bedrock aquifer underlying the Bishop Tube site. The field investigative procedures for each project task are outlined in the following sections. Representatives from the PADEP Southeastern Regional Office were present during a portion of the field investigation activities. Decisions regarding the sampling locations and the necessary analytical parameters for the groundwater samples were made collaboratively by Baker and the PADEP representatives, taking into consideration the project objectives and field conditions.

2.2.2   Installation of Additional Monitoring Wells and Packer Testing

The existing groundwater monitoring well network at the Bishop Tube site after the completion of the Phase I and Phase II investigation work included wells less than 100 feet in depth. The analytical results for the groundwater samples collected previously from deep bedrock monitoring wells MW17 and MW19 (both now abandoned) were found to contain concentrations of TCE exceeding 100,000 µg/l. TCE exhibits a specific gravity greater than that of water. Due to its higher specific gravity, TCE exhibits the tendency to sink within an aquifer. To more fully evaluate changes in water quality with respect to depth within the underlying fractured bedrock aquifer, four multi-screened monitoring wells were drilled and installed at the Bishop Tube site. Baker retained the services of a subcontractor (Uni-Tech Drilling

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Company, Inc.) to drill and install the four new multi-screened monitoring wells, as well as to perform hydraulic tests to more fully evaluate the subsurface conditions. The procedures for performing the drilling and testing are outlined in the following paragraphs.

2.2.2.1 Installation of Monitoring Wells

Based upon discussions with the PADEP Project Officer and considering the results of the previous Phase I and Phase II investigations (Baker 2002a, 2002b, 2003), four multi-screen monitoring wells were drilled in the following locations at the Bishop Tube site: 1) one multi-screened well (MW25) was drilled along the northern edge of Building #8, in the general vicinity of monitoring wells MW02 and MW03 (intended to replace monitoring well MW19 that was abandoned in February 2001); 2) a second multi-screened well (MW26) was drilled at the northeast corner of the property, in the vicinity of monitoring wells MW08 and MW09 (intended to evaluate the concentrations of chlorinated solvents near the downgradient edge of the property); 3) a third multi-screened monitoring well (MW27) was drilled within the former drum storage area, near monitoring well MW22 (intended to evaluate the concentrations of chlorinated solvents contained within the deeper portions of the fractured bedrock aquifer underlying this source area); and 4) a fourth multi-screened monitoring well (MW28) was drilled offsite within the Norfolk Southern Railroad right-of-way, in the general vicinity of monitoring wells MW15 and MW16 (intended to be a replacement for monitoring well MW17 that was abandoned in February 2001). The drilling locations for the multi-screened monitoring wells are shown on Figure 2.

To prevent the downward migration of TCE to deeper levels within the fractured bedrock aquifer, two strings of steel casing materials were used to construct the monitoring wells. Packer testing was performed at selected intervals during the drilling procedures to evaluate both changes in the concentration of TCE with depth and the hydraulic properties of the water-bearing fracture zones. In addition, geophysical well survey techniques were used to identify the depth of the water-bearing fractures in the borehole of each multi-screened monitoring well.

Information obtained from previous investigations (i.e., consultant reports prepared for the Christiana Metals Corporation) and by Baker on behalf of the PADEP found that the soils underlying the former vapor degreaser area in Building #8 and the former drum storage area contain relatively high amounts of TCE (i.e., greater than 4,000,000 μg/kg). This information coupled with the results of groundwater samples collected from the former and existing monitoring wells at the site suggested that separate-phase DNAPLs may locally exist in both the soils and in the underlying fractured bedrock aquifer. To avoid drilling the boreholes of the new monitoring wells through pools of separate-phase DNAPLs that may allow chlorinated solvents to migrate deeper within the underlying fractured bedrock aquifer, FLUTe® NAPL liners were deployed in the shallow portions of the boreholes of monitoring wells MW25 and MW27 to determine the presence/absence of DNAPLs in the overburden materials. FLUTe® NAPL liners were also deployed within the open borehole sections of monitoring wells MW25 and MW27 to determine the presence/absence and the depth of occurrence of potential DNAPLs. During the drilling operations, Baker periodically screened the soil and drill cuttings by testing these materials using a photoionization detector (PID), a combustible gas indicator (CGI), and Oil-Red-O dye. No free phase DNAPLs were observed in the soils and rock cuttings generated during the drilling operations for the new monitoring wells at the site.

2.2.2.2 Drilling Procedures for Multi-screened Monitoring Wells MW25 and MW27

For evaluating the potential occurrence of DNAPLs in the soils and weathered bedrock materials, the boreholes for monitoring wells MW25 and MW27 were first drilled using 6-inch diameter hollow-stem augers. The boreholes were drilled to the top of rock. Upon encountering the top of rock, the augers were removed from each borehole. To evaluate the presence/absence and the potential depth of DNAPLs occurring within the overburden materials, a FLUTe® NAPL liner was deployed in the annular space of each borehole. The FLUTe® NAPL liners were deployed in accordance with the procedures recommended by the Flexible Liner Underground Technologies Ltd. Company (see Appendix A).

To perform the packer testing procedures, geophysical well logging surveys, and the deployment of the FLUTe® NAPL liners in the open borehole sections of monitoring wells MW25 and MW27, a nominal 10-inch diameter hole was drilled to a depth of 16 feet and 39 feet, respectively, below the ground surface using air rotary techniques. Temporary 6-inch diameter casing was installed within the open section of each borehole. A nominal 6-inch diameter borehole was then drilled using air rotary techniques to a target depth of 280 feet below the ground surface for monitoring well MW25, and 250 feet below the ground surface for monitoring well MW27. Following the drilling of the 6-inch diameter boreholes, geophysical well logging techniques were used to log the open section of each well. To evaluate changes in the concentration of TCE with respect to depth and the hydraulic properties of the water-bearing zones encountered in each borehole, packer testing was performed at selected intervals using the information provided by the geophysical well logging results and the drilling logs. Six straddle-packer tests were performed in the 6-inch diameter borehole for monitoring well MW25. Eight straddle-packer tests were performed in the borehole for monitoring well MW27. Baker collected one groundwater sample from each yielding packer test interval for evaluating changes in the concentration of TCE with respect to depth in the aquifer (i.e., total of five groundwater samples from each borehole). These groundwater samples were sent to Analytical Laboratory Services, Inc., a private testing laboratory contracted by Baker for overnight screening analysis of selected chlorinated VOCs (i.e., 1,1-DCE, 1,2-DCE, 1,1,1-TCA, TCE, PCE, and vinyl chloride). To evaluate chemical changes of the groundwater with respect to depth in the aquifer, Baker measured the pH, specific conductance, and temperature of the groundwater samples collected during each packer test. A summary of the packer test intervals, groundwater sampling depths, groundwater yields, and water quality indicator parameters measured during the packer tests performed in the boreholes of monitoring wells MW25 and MW27 is presented in Table 1. At the conclusion of the packer testing procedures, the borehole of each well was cleaned and developed by circulating the drill rig compressed air. This well development process was performed until the groundwater appeared clear of sediment and suspended solids.

Following the conclusion of the packer testing procedures, FLUTe® NAPL liners were deployed in the open borehole section of monitoring wells MW25 and MW27. The FLUTe® NAPL liners were deployed in accordance with the procedures recommended by the FLUTe Ltd. Company (see Appendix A). Upon completion of the FLUTe® NAPL liner testing, the temporary 6-inch diameter steel casing was removed from the top of each borehole. The upper portion of each borehole was then reamed out using a nominal 15-inch diameter air rotary drill bit. Permanent 12-inch diameter steel casing was installed to a depth of 25.5 feet in the borehole of monitoring well MW25 and to a depth of 50 feet in the borehole of monitoring well MW27. The 12-inch steel casing was pressure grouted in place using a mixture of Portland cement and 5% bentonite. A nominal 12-inch diameter drill bit was then used to ream out the boreholes of monitoring wells MW25 and MW27 to a depth of 100 feet and 70 feet, respectively, using

air rotary drilling techniques. Permanent 8-inch diameter steel casing was installed within the borehole of monitoring well MW25 to a depth of 100 feet and in the borehole of monitoring well MW27 to a depth of 70 feet. The 8-inch steel casing was pressure grouted in place using a mixture of Portland cement and 5% bentonite. The boreholes for monitoring wells MW25 and MW27 were then reamed out to a depth of 282 and 242 feet, respectively, using a nominal 8-inch diameter air rotary drill bit.

The segregation of the boreholes for monitoring wells MW25 and MW27 into the three multi-screened sections began following the completion of the drilling activities. The information obtained from the analysis of the groundwater samples collected during the packer tests, the geophysical well logging procedures, and the deployment of the FLUTe® NAPL liners was used collectively for selecting the depth(s) and length(s) of the well screen intervals. Three separate intervals (i.e., shallow, intermediate, and deep) were screened within the boreholes of monitoring wells MW25 and MW27. Due to the nature and concentrations of the organic compounds dissolved in the groundwater, each screened interval was constructed using 2-inch diameter Schedule 5 stainless steel (type 304) wire-wrapped 20-slot well screen. To reduce well construction costs, 2-inch diameter Schedule 40 PVC casing materials were used to seal the intervals from the top of the stainless steel well screen to the ground surface. The well screen intervals were backfilled with #2 Morie sand. Bentonite was used to seal/isolate each screened interval within the boreholes. Each multi-screened monitoring well was completed at the surface using a +2-foot stick-up of 8-inch diameter steel casing with a locking well cap flange. A summary of the construction of monitoring wells MW25 and MW27 is outlined in Table 2.

2.2.2.3 Drilling Procedures for Multi-screened Monitoring Wells MW26 and MW28

To perform the packer testing and geophysical well logging procedures in the boreholes for monitoring wells MW26 and MW28, a nominal 10-inch diameter hole was drilled to a depth of 31 feet and 26 feet, respectively, below the ground surface using air rotary techniques. Temporary 6-inch diameter casing was installed within the open section of each borehole. A nominal 6-inch diameter borehole was then drilled using air rotary techniques to a target depth of 250 feet for monitoring well MW26 and a target depth of 252 feet for monitoring well MW28. Following the drilling of the 6-inch diameter boreholes, geophysical well logging techniques were used to log the open section of each well.

To evaluate changes in the concentration of TCE with respect to depth and the hydraulic properties of the water-bearing zones encountered in each borehole, packer testing was performed at selected intervals using the information provided by the geophysical well logging results and the drilling logs. A total of nine straddle-packer tests were performed in the 6-inch diameter borehole of monitoring well MW26. Baker collected a total of five groundwater samples from the yielding packer test intervals of monitoring well MW26 for evaluating changes in the concentration of TCE with respect to depth in the aquifer. A total of seven straddle-packer tests were performed in the borehole of monitoring well MW28. Baker collected a total of five groundwater samples from the yielding packer test intervals of monitoring well MW28 for evaluating changes in the concentration of TCE with respect to depth in the aquifer. The groundwater samples were sent to Analytical Laboratory Services, Inc., a private testing laboratory contracted by Baker for overnight screening analysis of selected chlorinated VOCs (i.e., 1,1-DCE, 1,2-DCE, 1,1,1-TCA, TCE, PCE, and vinyl chloride). To evaluate chemical changes of the groundwater with respect to depth in the aquifer, Baker measured the pH, specific conductance, and temperature of the groundwater samples collected during each packer test. A summary of the packer test intervals, groundwater sampling depths, groundwater yields, and water quality indicator parameters measured during the packer tests performed in the boreholes of monitoring wells MW26 and MW28 is presented in

Table 1. At the conclusion of the drilling and packer testing procedures, the borehole of each well was cleaned and developed by circulating the drill rig compressed air. This well development process was performed until the groundwater appeared clear of sediment and suspended solids.

Following the conclusion of the packer testing procedures, the temporary 6-inch diameter steel casing was removed from the top of each borehole. The upper portion of the borehole for monitoring well MW26 was reamed out using a nominal 15-inch diameter air rotary drill bit from the ground surface to a depth of 31.5 feet. For monitoring well MW28, the upper portion of the borehole was reamed out using a nominal 15-inch diameter air rotary drill bit from the ground surface to a depth of 26 feet. Permanent 12-inch diameter steel casing was inserted into the boreholes of monitoring wells MW26 and MW28 to a depth of 31.5 feet and 26 feet, respectively. The 12-inch diameter steel casing for each of these monitoring wells was then pressure grouted in place using a mixture of Portland cement and 5% bentonite. A nominal 12-inch diameter drill bit was then used to ream out the boreholes of monitoring wells MW26 and MW28 to a depth of 85 feet and 95 feet, respectively, using air rotary drilling techniques. Permanent 8-inch diameter steel casing was installed within the borehole of monitoring well MW26 to a depth of 85 feet and in the borehole of monitoring well MW28 to a depth of 95 feet. The 8-inch steel casing was pressure grouted in place using a mixture of Portland cement and 5% bentonite. The boreholes for monitoring wells MW26 and MW28 were then reamed out to a depth of 250 and 252 feet, respectively, using a nominal 8-inch diameter air rotary drill bit.

The segregation of the boreholes for monitoring wells MW26 and MW28 into the three multi-screened sections began following the completion of the drilling activities. The information obtained from the analysis of the groundwater samples collected during the packer tests and the geophysical well logging procedures was used collectively for selecting the depth(s) and length(s) of the well screen intervals. Three separate intervals (i.e., shallow, intermediate, and deep) were screened within the boreholes of monitoring wells MW26 and MW28. Due to the nature and concentrations of the organic compounds dissolved in the groundwater, each screened interval was constructed using 2-inch diameter Schedule 5 stainless steel (type 304) wire-wrapped 20-slot well screen. To reduce well construction costs, 2-inch diameter Schedule 40 PVC casing materials were used to seal the intervals from the top of the stainless steel well screen to the ground surface. The well screen intervals were backfilled with #2 Morie sand. Bentonite was used to seal/isolate each screened interval within the boreholes. Each multi-screened monitoring well was completed at the surface using a +2-foot stick-up of 8-inch diameter steel casing with a locking well cap flange. A summary of the construction of monitoring wells MW26 and MW28 is outlined in Table 2.

2.2.3   Geophysical Well Logging

To obtain information regarding the depth of the water-bearing fractures, variations in water quality, and changes in lithology, the following geophysical borehole logging techniques were performed in the boreholes of the four new multi-screened monitoring wells at the site. In addition, the geophysical borehole logging techniques were also used to evaluate the subsurface materials and the construction of the well at 30 Conestoga Road:

- Fluid Temperature Survey
- Fluid Conductivity Survey
- Caliper Survey

- Spontaneous Potential Survey
- Resistivity Survey
- Natural Gamma Ray Survey

The temperature and fluid conductivity surveys were performed first in each well, followed by the natural gamma ray, spontaneous potential, resistivity suite, and caliper log.

A temperature survey provides information regarding thermal changes of the groundwater and rock materials within the environment of the well. Abrupt changes in temperature provide an indication of zones where water may be either entering or exiting the borehole. The fluid conductivity survey provides information regarding the chemical activity (i.e., electrolytic properties) of the groundwater contained in the well. Groundwater containing relatively low concentrations of dissolved solids will exhibit low conductivity readings. Conversely, groundwater that contains elevated concentrations of dissolved solids will exhibit high conductivity readings. The measurements recorded during a fluid conductivity survey provide information regarding relative chemical changes in groundwater quality, and the location of zones where water is either entering or exiting the borehole. For the investigation conducted at the Bishop Tube site, the temperature and fluid conductivity surveys provided information regarding the depth of potential water-bearing fractures encountered by the borehole of each well, as well as indirect indication of water quality. These surveys were completed concurrently by lowering the logging probe within the groundwater contained in the wells at a slow and constant rate. The results were plotted on a combined temperature/conductivity log.

A caliper survey records changes in the diameter of a well with depth. This geophysical logging technique entails lowering an instrument to the bottom of the well, and then slowly pulling the sonde to the ground surface at a slow and constant rate. Spring-loaded extensions on the sides of the instrument rub against the edges of the borehole, and provide information regarding the changes in diameter of the well. The caliper survey is relatively sensitive to changes in the diameter of the borehole, and is useful for inferring the depth of potential fracture zones exposed along the open bedrock section of a well. The information obtained from the caliper survey was used in conjunction with the other log responses for designing the construction of the multi-screened monitoring wells. It should be noted that problems were encountered in the deployment of the caliper sonde within the boreholes of the monitoring wells. In each borehole poor results were obtained from the caliper survey. These problems are believed to be related to a deviation the boreholes (i.e., out-of-plumb) below a depth of 100 feet.

The geology of the area in the vicinity of the Bishop Tube site is complex. Published geologic information suggests that faults occur within the rocks of the area. To evaluate the characteristics of the deep portion of the fractured bedrock aquifer underlying the site, a suite of geophysical well logging devices (i.e., spontaneous potential, resistivity, and natural gamma ray) were used to survey the boreholes of monitoring wells MW25, MW26, MW27, and MW28, and the well at 30 Conestoga Road during the investigation. A spontaneous potential survey measures naturally occurring electrical potentials or voltages that may develop at the contacts between rocks of contrasting lithologies. The natural electric currents occurring in the borehole environment may be generated by one or a combination of the following effects: electrochemical differences; electrokinetic/streaming potentials; and/or oxidation/reduction potentials. Electrokentic effects are considered to be the most important phenomenon for generating the natural electric currents in the subsurface environment. These currents are believed to be produced by the migration of ions from a concentrated electrolytic solution to a dilute electrolytic

solution. The migration of these ions is strongly affected by the presence of both clays and shales, which function to decrease the mobility of negatively charged ions (i.e., anions). A spontaneous potential survey was performed during the investigation to obtain information regarding zones of water loss or gain (i.e., streaming potential) within the boreholes of monitoring wells MW25, MW26, MW27, and MW28, and the well at 30 Conestoga Road. In addition, the spontaneous potential survey was used to obtain a qualitative indication of water salinity and rock type correlation/layer thickness.

A resistivity survey, commonly referred to as an electric log survey, provides information regarding the character and thickness of various strata encountered by the borehole of a well, as well as a relative indication of water quality by measuring the apparent resistivity of the surrounding rock materials and groundwater. In theory, a resistivity survey is conducted by lowering one or more electrodes into a borehole filled with fluid. An electric current is forced to flow from these electrodes or to other electrodes that may situated either in the borehole or at the ground surface near the top of the well. The electrical logging instrument then measures the current loss (i.e., resistance to flow) between the two electrodes. To obtain information regarding the physical properties of the rock materials, the following three resistivity surveys were conducted during the investigation: single point resistance, short normal resistivity, and long normal resistivity. A single point resistance survey is performed by suspending one current electrode in the well and placing another electrode at the ground surface. Current is directed down the borehole where it spreads out into the formation. Part of the current flows back to the surface electrode where the current drop is measured. The single point resistance survey provides relatively good vertical resolution of rock layers that exhibit contrasting physical properties. The short normal and long normal logs provide information regarding the resistivity of the rock materials by using two electrodes within the well. The spacing of the current electrodes suspended on the sonde instrument determines the depth of penetration into the surrounding rock materials. When the separation of the electrodes is 16 inches or less, the survey is referred to as a short normal log. When the spacing of the electrodes is 64 inches, the survey is referred to as a long normal log. For the investigation performed at the Bishop Tube site, the information provided by the resistivity logs was used to determine the physical characteristics and thickness of the rock strata, locating the tops and bottom of juxtaposed lithologies, and a relative indication of the water quality contained within the rock materials surrounding the boreholes of monitoring wells MW25, MW26, MW27, and MW28, and the well at 30 Conestoga Road.

A gamma ray survey measures the relative amount of naturally occurring radiation being emitted from the rock materials encountered by the borehole of the well. Certain radioactive elements occur naturally in igneous and metamorphic rocks and as allochthonous particles in sedimentary rocks. In general, clays and shales contain higher concentrations of radioactive particles than mature sandstones and carbonate rocks. Based on these general compositional relationships, the information furnished from a natural gamma ray survey can be used as a qualitative guide for stratigraphic correlation and the identification of rock types. The sonde instrument consists of a scintillation-type receiver and a counting circuit. Radiation intensity within the borehole is measured by the number of pulses detected by the instrument per unit of time. The intensity is generally expressed as the average number of counts per second or per minute. For the investigation performed at the Bishop Tube site, the natural gamma ray survey provided information to discern differences in the lithology of the rock materials encountered by the boreholes of monitoring wells MW25, MW26, MW27, and MW28, and the well at 30 Conestoga Road.

The spontaneous potential, resistivity (i.e., single point resistance, short normal, and long normal), and natural gamma ray logs are performed concurrently by lowering the instruments to the bottom of each

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well, and then pulling the sonde to the ground surface at a slow and constant rate. The readings recorded by the instruments were recorded electronically and then plotted out as a combined log.

Representatives from Baker were present during the geophysical well logging activities to direct Earth Data Northeast Inc.'s personnel to the survey locations and to ensure compliance with the site specific Health and Safety Plan.

2.2.4   Deployment of FLUTe® NAPL Liners in Selected Monitoring Wells

To evaluate the presence/absence of DNAPLs occurring within the overburden materials (i.e., soils) and fractures exposed along the edges of the boreholes for monitoring wells MW25 and MW27, oleophilic reactive FLUTe® NAPL liners were deployed in the annular space of each well.

As outlined above, the boreholes for monitoring wells MW25 and MW27 were first drilled to the top of bedrock using 6-inch diameter hollow-stem augers. Upon encountering the top of rock, the augers were removed from each borehole. To evaluate the presence/absence and the potential depth of separate-phase DNAPLs occurring within the overburden materials, a FLUTe® NAPL liner was deployed in the annular space of each borehole. A representative from FLUTe® (i.e., Carl Keller) visited the Bishop Tube site on December 9 and 10, 2002 to instruct Baker and the personnel of the well drilling subcontractor (Uni-Tech Drilling Company, Inc.) on the proper procedures for the installation and removal of the FLUTe® NAPL liners. The deployment of the FLUTe® NAPL liners was performed in accordance with the procedures recommended by FLUTe® (see Appendix A). It should be noted that because the overburden materials encountered within the boreholes of monitoring wells MW25 and MW27 were mainly unsaturated, the FLUTe® NAPL liners were installed and removed without having to remove water from the annular space.

Top of bedrock was encountered at a depth of 18 feet below the ground surface during the drilling of monitoring well MW25. The overburden section of monitoring well MW25 from the ground surface to a depth of 18 feet was tested using a FLUTe® NAPL liner. The FLUTe® NAPL liner was deployed in the borehole of monitoring well MW25 on December 9, 2002, and was allowed to sit in the ground overnight (i.e., +18 hours). The liner was removed on December 10, 2002. No staining indicative of separate-phase chlorinated solvents within the soils lining the edges of the borehole was observed on the FLUTe® NAPL liner removed from the borehole of monitoring well MW25.

Top of bedrock was encountered at a depth of 45 feet below the ground surface during the drilling of monitoring well MW27. The overburden section of monitoring well MW27 from the ground surface to a depth of 45 feet was tested using a FLUTe® NAPL liner. The FLUTe® NAPL liner was deployed in the borehole of monitoring well MW27 on December 9, 2002, and was allowed to sit in the ground for +3 hours. The liner was removed on December 9, 2002. No staining indicative of separate-phase chlorinated solvents within the soils lining the edges of the borehole was observed on the FLUTe® NAPL liner removed from the borehole of monitoring well MW27.

Following the drilling of the 6-inch diameter boreholes to the target depths for monitoring wells MW25 and MW27, FLUTe® NAPL liners were deployed in the open bedrock sections of these wells to evaluate the presence or absence of separate-phase chlorinated solvents (i.e., DNAPLs) within the fractures of the bedrock. The deployment of the FLUTe® NAPL liners was performed in accordance with the procedures recommended by FLUTe® (see Appendix A). It should be noted that because the boreholes of monitoring

wells MW25 and MW27 were filled with water, the FLUTe® NAPL liners were installed and removed using the recommended procedures for the removal and addition of water to the boreholes of the well (see Appendix A).

The borehole of monitoring well MW25 was drilled to a depth of 282 feet below the ground surface. The fractured bedrock section of monitoring well MW25 from 18 feet (bottom of temporary casing) to 282 feet was tested using a FLUTe® NAPL liner. The FLUTe® NAPL liner was deployed in the borehole of monitoring well MW25 on February 10, 2003, and was allowed to sit in the ground overnight (i.e., +18 hours). The liner was removed on February 11, 2003. Following the waiting period, the tether cord attached to the bottom of the NAPL liner was pulled upward to invert and extract the liner from the hole. Upon complete recovery at the ground surface, the liner was laid out on a clean piece of polyethylene sheet plastic. The reactive liner was then peeled off of the protective cover by sliding and re-inverting the liner over the exposed flexible tubing. An indelible marker was used to immediately mark the top and bottom of the portions of the liner. The liner was then split lengthwise using a knife/scissors and spread out on the polyethylene plastic for further inspection. Indications regarding the presence of separate-phase NAPLs show up on the liners as dark spots (i.e., staining) and/or a bleed through of the reactive dye on the outside cover of the Tyvek® material. To determine the depth of the NAPL fluids in the subsurface, a measuring tape was used to gauge the distance between the top of the liner (i.e., reference point at the ground surface) and the occurrence of the staining. These measurements were recorded in the notebook of the Baker field technician at the site. Following the inspection work, the liner was rolled up, placed in a clean plastic zip-lock bag, and labeled for future reference. It should be noted that trace amounts of staining (suggesting the presence of separate-phase DNAPLs) was observed on the reactive coating of the FLUTe® NAPL liner for monitoring well MW25 at a depth of 280 feet.

The borehole of monitoring well MW27 was drilled to a depth of 242 feet below the ground surface. The fractured bedrock section of monitoring well MW27 from 45 feet (bottom of temporary casing) to 242 feet was tested using a FLUTe® NAPL liner. The FLUTe® NAPL liner was deployed in the borehole of monitoring well MW27 on December 16, 2002, and was allowed to sit in the ground overnight (i.e., +18 hours). The liner was removed on December 17, 2002. Following the waiting period, the tether cord attached to the bottom of the NAPL liner was pulled upward to invert and extract the liner from the hole. Upon complete recovery at the ground surface, the liner was laid out on a clean piece of polyethylene sheet plastic. The reactive liner was then peeled off of the protective cover by sliding and re-inverting the liner over the exposed flexible tubing. An indelible marker was used to immediately mark the top and bottom of the portions of the liner. The liner was then split lengthwise using a knife/scissors and spread out on the polyethylene plastic for further inspection. Indications regarding the presence of separate-phase NAPLs show up on the liners as dark spots (i.e., staining) and/or a bleed through of the reactive dye on the outside cover of the Tyvek® material. To determine the depth of the NAPL fluids in the subsurface, a measuring tape was used to gauge the distance between the top of the liner (i.e., reference point at the ground surface) and the occurrence of the staining. These measurements were recorded in the notebook of the Baker field technician at the site. Following the inspection work, the liner was rolled up, placed in a clean plastic zip-lock bag, and labeled for future reference. It should be noted that, no staining indicative of separate-phase chlorinated solvents within the fractured bedrock lining the edges of the borehole was observed on the FLUTe® NAPL liner removed from the borehole of monitoring well MW27.

2.2.5   Collection of Quarterly Groundwater Samples

To evaluate the concentrations of TCE within the fractured bedrock aquifer underlying the site, groundwater samples were collected from the monitoring wells on a quarterly basis for a period of one

year. The analytical results for the groundwater samples collected from the monitoring wells were compared to the Non-Residential Median Specific Concentrations listed in the *Pennsylvania Land Recycling Regulations*, dated November 24, 2001. This comparison provided information to assess the existing groundwater quality conditions, and further options for investigating/remediating the site.

To assess the groundwater conditions at the site, an initial round of groundwater samples was collected from each monitoring well (onsite and offsite locations) in March/April 2003. This initial sampling event included the sampling of 33 monitoring wells at the site, the well at 30 Conestoga Road, and the private water supply well located at 54 Conestoga Road. The groundwater samples collected from the private water supply well located at 54 Conestoga Road were collected from the pre, middle, and post sample ports installed on the water treatment system in this residential home. The groundwater samples collected during the March/April 2003 investigation were submitted to Severn Trent Laboratories, Inc. (i.e., the selected state contract laboratory) for analysis of TCL VOCs (including methyl tertiary butyl ether), total concentrations of TAL metals, carbonate and bicarbonate alkalinity, chloride, sulfate, sulfide, fluoride, methane, nitrate, nitrite, total nitrogen, total dissolved solids, and total organic carbon (see Table 3.1). To evaluate the potential onsite migration of hydrocarbon compounds, the groundwater samples collected from monitoring well MW24 (located by the entrance road near the Cooling Tower) were also analyzed for SVOCs.

A second round of groundwater samples was collected in July 2003 from the 33 monitoring wells at the site and the well at 30 Conestoga Road. In addition, discrete top and bottom samples were collected from 27 monitoring wells and the well located at 30 Conestoga Road. The groundwater samples collected during this sampling event were submitted to Lionville Laboratory, Inc. (i.e., the selected state contract laboratory) and analyzed for the following parameters: TCL VOCs (including methyl tertiary butyl ether and 1,4-dioxane), total and dissolved concentrations of TAL metals, carbonate and bicarbonate alkalinity, chloride, fluoride, methane, ethane, ethene, nitrate, nitrite, total nitrogen, sulfate, sulfide, total dissolved solids, and total organic carbon (see Table 3.2). To evaluate the potential onsite migration of hydrocarbon compounds, the groundwater samples collected from MW24 (located by the entrance road near the Cooling Tower) also were analyzed for SVOCs. The groundwater samples collected for analysis of dissolved metals were filtered in the field using an in-line 0.45-micron disposable filter connected to the pump discharge hose. A new filter was used for each well, as well as for each QA/QC sample.

The groundwater samples collected during the July 2003 sampling event were also analyzed for 1,4-dioxane. This compound was commonly used as a stabilizer for other chlorinated solvents including 1,1,1-TCA, PCE, and TCE. 1,4-dioxane is a highly soluble and mobile VOC. Based upon its high solubility, 1,4-dioxane tends to migrate in groundwater at the front of chlorinated solvent plumes. Based upon the large amounts of chlorinated solvents used at the Bishop Tube site, the groundwater samples collected from the multi-screened monitoring wells MW25, MW26, MW27, and MW28, and the well located at 30 Conestoga Road were analyzed for 1,4-dioxane. Although considered a VOC, 1,4-dioxane is generally measured by the testing laboratory using method 8270C (i.e., semi-volatile analysis). The use of semi-volatile testing methods for analyzing groundwater samples suspected to contain 1,4-dioxane allows for lower minimum detection limits to be obtained.

The third round of groundwater samples was collected in October 2003 from the 33 monitoring wells at the site, the well at 30 Conestoga Road, and the private water supply well located at 54 Conestoga Road. The groundwater samples collected from the private water supply well located at 54 Conestoga Road were collected from the pre, middle, and post sample ports installed on the water treatment system in this residential home. In addition, pre-purging and post-purging discrete bottom samples were collected from

monitoring well MW26. The groundwater samples collected during this sampling event were submitted to Lionville Laboratory, Inc. (i.e., the selected state contract laboratory) and analyzed for the following parameters: TCL VOCs (including methyl tertiary butyl ether), total concentrations of TAL metals, carbonate and bicarbonate alkalinity, chloride, fluoride, methane, ethane, ethene, nitrate, nitrite, total nitrogen, sulfate, sulfide, total dissolved solids, and total organic carbon (see Table 3.3). To evaluate potential DNAPL saturations within the fractured bedrock, groundwater samples were collected from monitoring wells MW01, MW02, MW22, MW23, MW25A, MW25B, MW26B, MW28C, and the well at 30 Conestoga Road for the analysis of radon. The groundwater samples collected for the measurement of radon were submitted to the PADEP Bureau of Laboratories, located in Harrisburg, Pennsylvania for analysis.

A fourth round of groundwater samples was collected in February 2004 from the 33 monitoring wells at the site and the well at 30 Conestoga Road. In addition, discrete bottom samples were collected from 27 monitoring wells and the well located at 30 Conestoga Road. The groundwater samples collected during this sampling event were submitted to Lionville Laboratory, Inc. (i.e., the selected state contract laboratory) and analyzed for the following parameters: TCL VOCs (including methyl tertiary butyl ether), total and dissolved concentrations of TAL metals, carbonate and bicarbonate alkalinity, chloride, fluoride, methane, ethane, ethene, nitrate, nitrite, total nitrogen, sulfate, sulfide, total dissolved solids, and total organic carbon (see Table 3.4). The groundwater samples collected for analysis of dissolved metals were filtered in the field using an in-line 0.45-micron disposable filter connected to the pump discharge hose. A new filter was used for each well, as well as for each QA/QC sample. To evaluate potential DNAPL saturations within the fractured bedrock, groundwater samples were collected from monitoring wells MW02, MW03, MW12, MW25A, MW25B, MW25C, MW26A, MW26B, MW26C, MW28A, MW28B, and MW28C for the analysis of radon. The groundwater samples collected for the measurement of radon were submitted to the PADEP Bureau of Laboratories, located in Harrisburg, Pennsylvania for analysis.

To reduce the volume of water to be purged from the wells, low-flow purging and sampling techniques were used during the investigation to collect the groundwater samples from the monitoring wells. This sampling procedure involved setting a submersible pump adjacent to a water-bearing fracture zone. The information recorded during the well drilling procedures and/or from the well logs was used to identify the location of the primary water-bearing fracture within the borehole/screened interval of each well, as well as to select the depth of the submersible pump. Each monitoring well was purged at a relatively low pumping rate (i.e., less than one liter per minute). Static water levels were measured on a periodic basis during the purging process to assure drawdown in the well was minimized. The groundwater pumped from the monitoring wells during the low-flow purging procedures was monitored for water quality indicator parameters (i.e., pH, temperature, specific conductance, dissolved oxygen, and redox potential) to determine when equilibrium conditions were reached (i.e., ± 10% over a three to ten minute period) at which time groundwater samples were collected from each monitoring well. A summary of the final water quality indicator parameters recorded prior to collecting the groundwater samples in each monitoring well is presented in Tables 4.1 through 4.8. If the well purged dry, it was allowed to recharge. If full recovery was not achieved within two hours, the groundwater samples were collected as water became available, filling the glassware containers for parameters in order of decreasing volatility.

During the July 2003 sampling event, Baker collected discrete top and bottom samples from selected monitoring wells at the site. Discrete bottom samples were also collected from selected monitoring wells during the February 2004 sampling event. Prior to sampling, each monitoring well was gauged using an electronic interface probe to determine the presence/absence of separate-phase DNAPL and LNAPL layers. Separate-phase DNAPL and/or LNAPL layers were not observed in any of the monitoring wells at

the site during the July 2003 sampling event. The samples collected from the groundwater/surface interface (i.e., LNAPL samples) in July 2003 were collected from monitoring wells MW04, MW05, MW06, MW08, MW09, and MW24. These discrete top samples were collected using a disposable bailer. A new bailer was used to collect the LNAPL samples from each well. To evaluate the static non-purge conditions, discrete samples were collected in July 2003 from the bottom intervals (i.e., DNAPL samples) of monitoring wells MW02, MW03, MW12, MW13, MW15, MW16, MW18, MW20, MW22, the multi-screened monitoring wells MW25, MW26, MW27, and MW28, and the well located at 30 Conestoga Road. Pre- and post-purge discrete samples were collected in February 2004 from the bottom intervals (i.e., DNAPL samples) of monitoring wells MW02, MW12, MW22, the multi-screened monitoring wells MW25, MW26, MW27B, and MW28, and the well located at 30 Conestoga Road. During the February 2004 sampling event, separate-phase DNAPLs were observed in monitoring wells MW25A, MW25B, MW26A, MW26B, MW27B, MW28A, MW28B, and MW28C. The DNAPL/bottom interval samples were collected during the July 2003, October 2003, and February 2004 sampling events using a discrete interval sampler (i.e., Solinst Model 425-T).

To minimize the amount of liquid IDW requiring disposal, the groundwater generated during the purging procedures was treated onsite by passing the raw influent through a 200-pound liquid-phase carbon adsorber. The used carbon was removed at the end of each quarterly groundwater sampling event and placed in a 55-gallon capacity steel DOT shippable drum. These drums of used carbon have been temporarily stored in the designated staging area at the site (i.e., next to the loading dock along the south side of Building #5). At the completion of the Groundwater Investigation, Baker will make arrangements with Waste Recovery Solutions, Inc. (i.e., approved IDW disposal subcontractor) for the load out, transportation, and disposal of the used activated carbon materials. New (i.e., virgin) carbon was placed in the liquid-phase adsorber prior to the start of each subsequent quarterly groundwater sampling event. During the well purging procedures, the treated effluent from the 200-pound liquid-phase carbon adsorber was discharged to the ground surface adjacent to each wellhead. At the end of each quarterly sampling event a groundwater sample was collected from the treated effluent to evaluate the efficiency of the liquid-phase adsorber in removing the VOCs contained in the raw influent.

New disposable polyethylene tubing was connected to the submersible pump and used for the collection of groundwater samples from each well. In addition, new polyethylene tubing was used for each sampling event in order to reduce the possibility of cross-contamination between wells and to eliminate the potential for the build-up and permeation of VOCs in the polyethylene plastic tubing materials if left in a well over the long-term. These procedures were instituted to provide more representative groundwater samples. All non-dedicated sampling equipment was decontaminated following the purging and collection of groundwater samples from each well. This included an Alconox® soap wash and deionized water rinse. The decontamination fluids generated during the sampling of the monitoring wells was temporarily containerized in 55-gallon capacity steel DOT shippable drums. To minimize IDW, the decontamination fluids were treated onsite by passing this water through the 200-pound liquid-phase carbon adsorber at the end of each quarterly groundwater sampling event. The treated decontamination water was discharged to the ground surface at the site.

2.2.6   Supplemental Site Survey

Baker retained the services of a Pennsylvania Registered Land Surveyor (i.e., Dawood Engineering, Inc.) to perform a supplemental survey of the site. The information gathered during the supplemental site survey work was used to prepare an inclusive site map of the Bishop Tube site.

The supplemental site survey work was performed on April 2, 2003 to establish the locations, ground elevations, and top of casing elevations for the four new monitoring wells (MW25, MW26, MW27, and MW28), as well as the well at 30 Conestoga Road. For these monitoring wells, the reference elevations were established at the top of PVC (where present) or steel casing.

The elevation measurements recorded by the subcontractor were tied into a control point in the vicinity of the Bishop Tube site for equating the data to mean sea level (i.e., North Geodetic Vertical Datum 1929). This control point consists of a nail set in B.T. CO utility pole #5 (elevation 390.5 feet) located in the northeast corner of the Bishop Tube property (see Figure 2). Representatives from Baker were present during the surveying activities to direct the survey subcontractor the points and features to be mapped at the site.

The new monitoring well locations established during the supplementary site survey were incorporated into the existing survey database to create one composite map of the area. The site map included the following information: pertinent site features (i.e., buildings, fencing, gravel covered areas, asphalt covered areas, property boundaries, new monitoring wells, and surface topography). The electronic data produced from the survey activities were used to develop a site map using AutoCAD™ Version No. 2000 software.

2.2.7   Management of IDW Materials

Investigation Derived Waste (IDW) materials generated during the investigation activities were managed appropriately to minimize exposure to potential substances of environmental concern, and impacts to human health and the environment.

The analytical results for the soil samples collected previously at the site show that the subsurface materials underlying the areas where the new monitoring wells were drilled contain elevated concentrations of chlorinated solvents. Based on this information, the cuttings and groundwater generated during the installation of MW25, MW26, MW27, and MW28 were managed appropriately to protect human health and the environment. At each drilling location, a 20 cubic yard sealed bottom roll-off container was used to manage the drill cuttings and groundwater generated during the drilling operations. The rock cuttings and groundwater generated during the drilling procedures were directed to the roll-off container via a wellhead diverter and flow line.

The drilling subcontractor was responsible for periodically pumping or using a smaller tank to transfer the groundwater within the roll-off containers to an 18,000-gallon capacity compartmented frac tank that was staged at the site (i.e., along the east side of Building #8). To further minimize the amount of IDW requiring disposal, the groundwater generated during the development of the monitoring wells and the wash waters produced from the decontamination of the drilling equipment were also transferred to the 18,000-gallon capacity frac tank. More than 100,000-gallons of groundwater were generated during the drilling, well development, and equipment clean up procedures during the investigation. Based upon the volume of water generated during the well drilling work, personnel from Baker periodically pumped out the fluids contained in the frac tank, passing the raw groundwater through a sediment filter and a 1,000-pound liquid-phase carbon adsorber. The treated groundwater was ultimately discharged to the ground surface along the eastern property boundary of the site. To verify that the 1,000-pound liquid-phase carbon adsorber was effectively reducing the concentrations of VOCs to levels below the PADEP Non-Residential Statewide Health-based Groundwater Standards, Baker collected treated effluent samples at

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the end of each discharge event. These groundwater samples were submitted to a private testing laboratory (i.e., Analytical Laboratory Services, Inc.) for rapid (i.e., overnight) screening analysis of VOCs. The analytical results for the treated groundwater samples showed that the concentrations of chlorinated solvents were less than the PADEP Non-Residential Statewide Health-based Groundwater Standards (see Tables 5.1 and 5.2).

The groundwater pumped from the roll-off containers to the frac tank contained silt and other suspended solids. Over time these silts and fines settled to the bottom of the frac tank. Per the agreement with the rental company (i.e., Rain-For-Rent) who furnished the 18,000-gallon capacity frac tank, Baker was required to remove the silts and sediments that had accumulated in the vessel prior to its return. Entry into the tank was required to remove the silt and sediments. Due to limited access, the inside of the frac tank was considered a confined space. Accordingly, Baker retained the services of a subcontractor (i.e., TIER DE, Inc.) to provide personnel and equipment to remove the silts and fines from the bottom of the frac tank. The subcontractor furnished personnel trained in confined space entry, as well as the necessary support personnel, equipment (i.e., vacuum truck), safety equipment, and a standby emergency rescue team necessary for performing work in a confined space. All work was performed in accordance with 29CFR 1910.146 (dealing with entry and work in a confined space). The clean out of the frac tank occurred on May 14, 2003. The silt and fines removed from the tank were containerized in a 2,000-gallon capacity vacuum truck. In addition, at the time the frac tank was cleaned out, the subcontractor vacuumed out the used carbon contained within the 1,000-pound liquid-phase carbon adsorber. The clean out of the frac tank and the 1,000-pound liquid-phase adsorber was performed using OSHA Level C PPE.

Prior to the start of the well drilling procedures, contracting arrangements were made by Baker with Waste Recovery Solutions, Inc. of Myerstown, Pennsylvania for the load out, transportation, and disposal of the drill cuttings, sludges contained in the frac tank, and the used liquid-phase carbon materials. Since Waste Recovery Solutions, Inc. had previously handled similar waste streams in March of 2002 for the Bishop Tube site no additional sampling or paperwork was necessary to characterize the waste materials for disposal.

During the drilling operations, a cumulative total of 59.69 tons of soil/drill cuttings were generated. The soils/drill cuttings were transported from the Bishop Tube site by Waste Recovery Solutions, Inc. and ultimately disposed of at the Waste Management, Inc., Modern Landfill facility, located in York County, Pennsylvania (PADEP Municipal and Residual Waste Processing Permit No. 100113). Waste Recovery Solutions, Inc. picked up the soil/drill cuttings at the Bishop Tube site and delivered these IDW materials to the Modern Landfill facility on January 6, 2003, January 16, 2003, February 25, 2003, March 4, 2003, and March 20, 2003. A copy of the certificates documenting the disposal of the soil/drill cuttings is included in Appendix B.

A combined total of 15.15 tons of sludge/water and used carbon were removed from the 18,000-gallon capacity frac tank and 1,000-pound adsorber by TIER DE, Inc. on May 14, 2003. The sludges/used carbon materials were transported from the Bishop Tube site by TIER DE, Inc. to Waste Recovery Solutions, Inc.'s facility in Myerstown, Pennsylvania on May 15, 2003. Waste Recovery Solutions, Inc. solidified these waste materials at their Myerstown, Pennsylvania facility by mixing the sludges/used carbon with sawdust. The solidified sludges/used carbon materials were transported by Waste Recovery Solutions, Inc. to the Waste Management, Inc., Modern Landfill facility, located in York County, Pennsylvania (PADEP Municipal and Residual Waste Processing Permit No. 100113) for disposal on

May 16, 2003. A copy of the certificate documenting the disposal of the sludges/used carbon materials is included in Appendix B.

During the collection of the quarterly groundwater samples in March/April 2003, July 2003, October 2003, and February 2004, a 200-pound liquid phase carbon adsorber was used to treat the water purged from the monitoring wells. At the end of each of these sampling events, the used carbon materials were placed in a separate 55-gallon capacity steel DOT shippable drum. These drums were staged next to the loading dock along the south side of Building #5 at the site. Waste Recovery Solutions, Inc picked up the four drums containing the used carbon materials at the Bishop Tube site on May 3, 2004. The used carbon materials were ultimately disposed of at the Waste Management, Inc., Modern Landfill facility, located in York County, Pennsylvania (PADEP Municipal and Residual Waste Processing Permit No. 100113) on May 10, 2004. A copy of the certificate documenting the disposal of these used carbon materials is included in Appendix B.

2.2.8   Development of Groundwater Flow Model

To evaluate the regional direction(s) of groundwater flow and to estimate the rate of TCE migration within the fractured bedrock aquifer, a three-dimensional groundwater flow model was developed by Baker for the area of Chester Valley surrounding the Bishop Tube site. The carbonate-rich rocks in the Chester Valley supply a large amount of water to businesses and residents within the region. The water quality of many drinking water supplies within the watershed has been degraded by chlorinated solvents (i.e., TCE). Importantly the information provided by the groundwater flow model was used to assess potential receptors situated hydraulically downgradient to the site. Specific objectives for the modeling effort included:

- Evaluating the existing groundwater elevation and chemical data.
- Developing an internally consistent conceptual model in order to design a three-dimensional numerical groundwater flow model.
- Constructing the flow model with reasonable assumptions.
- Calibrating the flow model to observed groundwater elevations.
- Estimating the distance the groundwater plume containing chlorinated solvents has traveled.
- Providing a re-usable tool for future site management efforts.

The modeling effort consisted of generating a groundwater model flow using MODFLOW-2000 and MODPATH computer programs. Contaminant transport modeling was evaluated using the computer program RT3D. The modeling effort was broken into the following four tasks: 1) data review and model setup; 2) groundwater flow modeling; 3) solute transport modeling; and 4) reporting. The solute transport model provided a prediction of the groundwater plume behavior over time. Based upon the results, it was possible to estimate the time required for TCE and its degradation products to migrate to nearby potential receptors.

Baker used GMS version 4.0 (ERML, 2003) and MODFLOW-2000 (Harbaugh et al., 2000) program codes to construct the flow model. After the flow model was calibrated, the transport models were constructed using RT3D (Clement, 1997, 1999) atop the groundwater flow field generated by MODFLOW-2000. Simulations of groundwater flow in fractured bedrock are not usually performed using a model that assumes flow in porous media; however, at the scale of the model, the network of fractures is sufficiently well connected such that it behaves as an *equivalent porous medium* (EPM). It

should be noted, that as the scale of the simulations increase, the error associated with the simulation decreases; that is, accuracy cannot be expected for a simulation on the order of tens of feet from the site, but increases as the distance from the site increases to hundreds or thousands of feet. Accordingly, the use of MODFLOW-2000 is reasonable at the scale of hundreds or thousands of feet from the site. Sloto (1990) used an earlier version of MODFLOW to simulate groundwater flow in Chester Valley.

The model grid was divided into four layers, with each successive underlying layer having a hydraulic conductivity value less than that above. The values were derived from the relationship between fracture density and depth (Sloto, 1990).

Three computer codes were used to predict the release and migration of chlorinated solvents at the Bishop Tube Site. The groundwater flow model MODFLOW-2000 was used to simulate groundwater flow. MODPATH was used to generate pathlines from the source areas over the modeled time period. The reactive transport model RT3D (Clement, 1997, 1999) was used to simulate the fate of trichloroethene (TCE) and its degradation products, cis-1,2- DCE and vinyl chloride (VC). The model simulations were conducted for a 52-year period from 1951 to 2003.

For the transport model, Baker assumed source locations and strengths as extrapolated from the available chemical data. The resulting locations lay within the footprint of the site property boundaries. Based upon the likely presence of separate-phase DNAPLs, the model was developed assuming that the source for the dissolved concentrations of chlorinated solvents is situated at depth within the fractured bedrock aquifer. A more detailed discussion regarding the set up of the groundwater flow model, the selection of input parameters, model calibration, and results are presented in Section 5.0 of the report.

2.3 **Sample Analytical Program**

The record keeping procedures and the laboratory testing methods used to analyze the environmental samples collected during the investigation are outlined in the following paragraphs.

2.3.1 **Introduction**

In accordance with the project objectives, the groundwater samples collected during the investigation were used to further characterize the concentrations of VOCs contained in the fractured bedrock aquifer underlying the site. The groundwater samples collected during the packer testing activities were submitted to Analytical Laboratory Services, Inc., Harrisburg, Pennsylvania for rapid overnight screening analysis (i.e., 24-hour turnaround) of selected VOCs. Analytical Laboratory Services, Inc. was retained by Baker as a separate subcontractor for the rapid turnaround analysis of the groundwater samples. The groundwater samples collected during the first quarterly groundwater sampling event in March/April 2003 were submitted for analysis to Severn Trent Laboratories, Inc. of Pittsburgh, Pennsylvania (i.e., PADEP selected state contract laboratory). The second, third, and fourth quarterly rounds of groundwater samples (collected in July 2003, October 2003, and February 2004, respectively) were submitted to Lionville Laboratory, Inc. of Lionville, Pennsylvania (i.e., PADEP selected state contract laboratory). The testing results for the groundwater samples analyzed by Severn Trent Laboratories, Inc. and Lionville Laboratory, Inc. followed CLP Type III reporting protocols.

Field Quality Assurance/Quality Control (QA/QC) samples were collected for each environmental medium of samples during the investigation. The QA/QC requirements for the selected sample groups

were performed in accordance with the guidelines outlined in the Quality Assurance Project Plan (QAPP) developed by Baker prior to initiation of the field investigation activities.

Eleven QA/QC samples were submitted to Severn Trent Laboratories, Inc. along with the groundwater samples collected from the monitoring wells at the site, the well at 30 Conestoga Road, and the private water supply well located at 54 Conestoga Road during the March/April 2003 sampling event. These QA/QC samples included two duplicate groundwater samples, two field blank samples, two rinsate samples, two matrix spike (MS) samples, two matrix spike duplicate (MSD) samples, and one effluent sample collected from the carbon adsorber. During transit to the testing laboratory, trip blank samples were placed in each cooler that contained samples for VOC analysis. A total of six trip blank samples were used during the March/April 2003 sampling event for documenting the sample handling procedures.

Eleven QA/QC samples were submitted to Lionville Laboratory, Inc. along with the groundwater samples collected from the monitoring wells at the site and the well located at 30 Conestoga Road during the July 2003 sampling event. These QA/QC samples included two duplicate groundwater samples, two field blank samples, two rinsate samples, and two MS samples, two MSD samples, and one effluent sample collected from the carbon adsorber. During transit to the testing laboratory, trip blank samples were placed in each cooler that contained samples for VOC analysis. A total of seven trip blank samples were used during the July 2003 sampling event for documenting the sample handling procedures.

Fourteen QA/QC samples were submitted to Lionville Laboratory, Inc. along with the groundwater samples collected from the monitoring wells, the well at 30 Conestoga Road, and the private water supply well located at 54 Conestoga Road during the October 2003 sampling event. These QA/QC samples included three duplicate groundwater samples, two field blank samples, two rinsate samples, three MS samples, three MSD samples, and one effluent sample collected from the carbon adsorber. During transit to the testing laboratory, trip blank samples were placed in each cooler that contained samples for VOC analysis. A total of seven trip blank samples were used during the October 2003 sampling event for documenting the sample handling procedures.

Fourteen QA/QC samples were submitted to Lionville Laboratory, Inc. along with the groundwater samples collected from the monitoring wells at the site and the well located at 30 Conestoga Road during the February 2004 sampling event. These QA/QC samples included two duplicate groundwater samples, two field blank samples, two rinsate samples, two MS samples, two MSD samples, and one effluent sample collected from the carbon adsorber. During transit to the testing laboratory, trip blank samples were placed in each cooler that contained samples for VOC analysis. A total of seven trip blank samples were used during the February 2004 sampling event for documenting the sample handling procedures.

In order to identify and accurately track the environmental samples collected during the investigation, including QA/QC samples, a unique number was given to each sample. This number was designed to provide information regarding the sample date, the sample media, sampling location, and QA/QC qualifiers. The sample designation format used during the investigation is as follows:

PADEP Site # – Sample Date – Medium – Sampling Location – QA/QC Designation

An explanation of each of these identifiers is given below.

PADEP Site #   116 (for all samples)

Sample Date   040103 – April 1, 2003

Medium   GW   – Groundwater
                        LNAPL/DNAPL   – NAPL matrix

Sampling Location   MW01   – Monitoring Well MW01
                        PT02   – Packer Test Sample Collected from the Second Depth Interval
                        30CR   – 30 Conestoga Road
                        54CR   – 54 Conestoga Road
                        EFF   – Treated Effluent Sample

QA/QC   The following designations were used for the QA/QC samples collected during the investigation:

                        DUP   – Duplicate Sample
                        RINSATE   – Equipment Blank (Rinsate) Samples
                        FIELD   – Field Blank Samples
                        MS/MSD   – Matrix Spike/Matrix Spike Duplicate Samples
                        TRIP   – Trip Blank Samples

Using this sample designation format the sample number 116-040103-GW-MW27B-DUP refers to:

116-040103-GW-MW27B-DUP   PADEP Site # for Bishop Tube Site
116-**040103**-GW-MW27B-DUP   Date collected – April 1, 2003
116-040103-**GW**-MW27B-DUP   Sample Type – Groundwater
116-040103-GW-**MW27B**-DUP   Sampling Location – Monitoring Well MW27B
116-040103-GW-MW27B-**DUP**   QA/QC Designation – Duplicate Sample

2.3.2   Analysis of Groundwater Samples Collected During Packer Testing Procedures

To evaluate changes in the concentration of TCE with respect to depth within the fractured bedrock aquifer underlying the Bishop Tube site, groundwater samples were collected during the packer testing procedures from each fractured interval that was found to yield water. The packer testing intervals were chosen based upon the observations made by the onsite geologist during the drilling procedures, the information provided by the geophysical well logs, and the results provided by the FLUTe® NAPL liner testing (only deployed in the boreholes of monitoring wells MW25 and MW27).

The packer testing groundwater samples were collected in 40-ml clear glass VOC vials fixed with hydrochloric acid (HCl) and secured with Teflon®-lined lids. The samples were collected directly from the water stream at the end of the pump discharge line. To preserve the integrity of the groundwater samples, the containers were stored in coolers at 4°C until picked up/delivered to Analytical Laboratory Services, Inc. for analysis. A total of five groundwater samples were collected during the packer testing procedures performed in the borehole of monitoring well MW25. During the packer testing procedures performed in the borehole of monitoring well MW26, Baker collected a total of five groundwater samples. A total of six groundwater samples were collected during the packer testing procedures performed in the borehole of monitoring well MW27. During the packer testing procedures performed in the borehole of monitoring well MW28, Baker collected a total of five groundwater samples.

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Challenge Us.

The groundwater samples collected during the packer testing procedures were submitted to a subcontractor (i.e., Analytical Laboratory Services, Inc.) retained by Baker for overnight analysis (i.e., 24-hour turnaround) of the following selected chlorinated VOCs: 1,1-DCE, cis-1,2-DCE, trans-1,2-DCE, PCE, 1,1,1-TCA, TCE, and vinyl chloride. Trip Blank samples were placed in each cooler during shipment of the packer testing groundwater samples to the testing laboratory. A summary of the analytical methods used to analyze the groundwater samples collected during the packer testing procedures is outlined in Table 6.

2.3.3   Analysis of Effluent Samples Collected from the Carbon Adsorber

The groundwater generated during the drilling of the additional monitoring wells was ultimately transferred to an 18,000-gallon capacity frac tank staged along the eastern side of Building #8. Baker periodically treated the groundwater contained in the frac tank during the course of the well drilling work by passing the raw water (suspected to contain elevated concentrations of chlorinated VOCs) through a 1,000-pound liquid-phase carbon adsorber. The treated effluent was discharged to the ground surface at the site. At the request of the PADEP Bureau of Water Quality, Baker periodically collected samples of the treated effluent to evaluate the effectiveness of the carbon adsorber in removing the chlorinated VOCs.

The treated effluent samples were collected in 40-ml clear glass VOC vials fixed with hydrochloric acid (HCl) and secured with Teflon®-lined lids. The samples were collected directly from the water stream at the bottom of the carbon adsorber. To preserve the integrity of the groundwater samples, the containers were stored in coolers at 4°C until picked up/delivered to Analytical Laboratory Services, Inc. for analysis. A total of five effluent samples were collected to monitor the concentrations of chlorinated VOCs dissolved in the treated groundwater.

The treated effluent samples were submitted to a subcontractor (i.e., Analytical Laboratory Services, Inc.) retained by Baker for overnight analysis (i.e., 24-hour turnaround) of the following selected chlorinated VOCs: 1,1-DCE, cis-1,2-DCE, trans-1,2-DCE, PCE, 1,1,1-TCA, TCE, and vinyl chloride. Trip Blank samples were placed in each cooler during shipment of the effluent samples to the testing laboratory. A summary of the analytical methods used to analyze the effluent samples collected during the investigation outlined in Table 6.

2.3.4   Analysis of Quarterly Groundwater Samples Collected from Monitoring Wells

To assess seasonal variations in the concentrations of organic and inorganic compounds contained in the fractured bedrock aquifer underlying the Bishop Tube site, groundwater samples were collected from the monitoring wells on a quarterly basis during the investigation. These quarterly groundwater sampling events were performed in March/April 2003, July 2003, October 2003, and February 2004.

During the March/April 2003 sampling event, groundwater samples were collected from a total of 35 wells (including all monitoring wells and the wells at 30 Conestoga Road and 54 Conestoga Road). In addition, eleven QA/QC samples were collected during the March/April 2003 sampling event. These QA/QC samples included two duplicate groundwater samples, two field blank samples, two rinsate samples, two matrix spike (MS) samples, two matrix spike duplicate (MSD) samples, and one effluent sample collected from the carbon adsorber. During transit to the testing laboratory, trip blank samples

were placed in each cooler that contained samples for VOC analysis. A total of six trip blank samples were used during the March/April 2003 sampling event for documenting the sample handling procedures.

The groundwater samples collected during the March/April 2003 sampling event were submitted to Severn Trent Laboratories, Inc. for analysis of TCL VOCs, TCL SVOCs (monitoring well MW24 only), total concentrations of TAL metals, bicarbonate alkalinity, carbonate alkalinity, chloride, sulfate, sulfide, fluoride, methane, nitrate, nitrite, total nitrogen, total dissolved solids, and total organic carbon. At the request of the PADEP, a single effluent sample was collected from the 200-pound carbon adsorber at the end of the March/April 2003 groundwater sampling event. This treated effluent water sample was submitted to the testing laboratory for analysis of TCL VOCs.

The March/April 2003 groundwater samples submitted to Severn Trent Laboratories, Inc. for TCL VOC analysis were collected in 40 mL clear glass VOC vials fixed with hydrochloric acid (HCl) and secured with Teflon®-lined lids. The groundwater samples submitted for the analysis of TCL SVOCs, total TAL metals, cyanide, nitrate, and water quality parameters were collected in 950 mL amber glass jars, 1 liter plastic containers fixed with HNO_3, 1 liter plastic containers fixed with NaOH, 250 mL plastic containers fixed with sulfuric acid (H_2SO_4), and 1 liter plastic containers with no preservatives, respectively. To preserve the integrity of the groundwater samples, the sample containers were stored in coolers at 4°C until delivered to the testing laboratory for analysis. A summary of the analytical methods used to analyze the groundwater samples collected from the monitoring wells during the March/April 2003 sampling event is outlined in Table 3.1.

During the July 2003 sampling event, groundwater samples were collected from a total of 33 wells (including all monitoring wells and the well at 30 Conestoga Road). Due to an unknown obstruction inside the casing, no groundwater samples were collected from monitoring well MW27 during the July 2003 sampling event. In addition, discrete top and bottom groundwater samples were collected from 27 of the monitoring wells and the well located at 30 Conestoga Road for analysis of TCL VOCs. To document the sample handling and equipment decontamination procedures, eleven QA/QC samples were collected during the July 2003 sampling event. These QA/QC samples included two duplicate groundwater samples, two field blank samples, two rinsate samples, two matrix spike (MS) samples, two matrix spike duplicate (MSD) samples, and one effluent sample collected from the carbon adsorber. During transit to the testing laboratory, trip blank samples were placed in each cooler that contained samples for VOC analysis. A total of seven trip blank samples were used during the July 2003 sampling event for documenting the sample handling procedures.

The groundwater samples collected during the July 2003 sampling event were submitted to Lionville Laboratory, Inc. for analysis of TCL VOCs, TCL SVOCs (monitoring well MW24 only), total and dissolved concentrations of TAL metals, bicarbonate alkalinity, carbonate alkalinity, chloride, sulfate, sulfide, fluoride, methane, ethane, ethene, nitrate, nitrite, total nitrogen, total dissolved solids, and total organic carbon. The groundwater samples collected from monitoring wells MW25A, MW25B, MW25C, MW26A, MW26B, MW26C, MW27A, MW27B, MW28A, MW28B, MW28C, and the well at 30 Conestoga Road were also submitted to the testing laboratory for the analysis of 1,4-dioxane. At the request of the PADEP, a single effluent sample was collected from the 200-pound carbon adsorber at the end of the July 2003 groundwater sampling event. This treated effluent water sample was submitted to the testing laboratory for analysis of TCL VOCs.

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The July 2003 groundwater samples submitted to Lionville Laboratory, Inc. for TCL VOC analysis were collected in 40 mL clear glass VOC vials fixed with hydrochloric acid (HCl) and secured with Teflon®-lined lids. The groundwater samples submitted for the analysis of TCL SVOCs, total and dissolved TAL metals, cyanide, nitrate, and water quality parameters were collected in 950 mL amber glass jars, 1 liter plastic containers fixed with HNO_3, 1 liter plastic containers fixed with NaOH, 250 mL plastic containers fixed with sulfuric acid (H_2SO_4), and 1 liter plastic containers with no preservatives, respectively. The samples collected for analysis of 1,4-dioxane were collected in two separate 950 mL amber glass jars. To preserve the integrity of the groundwater samples, the sample containers were stored in coolers at 4°C until delivered to the testing laboratory for analysis. A summary of the analytical methods used to analyze the groundwater samples collected from the monitoring wells during the July 2003 sampling event is outlined in Table 3.2.

During the October 2003 sampling event, groundwater samples were collected from a total of 34 wells (including all monitoring wells and the wells at 30 Conestoga Road and 54 Conestoga Road). Due to an unknown obstruction inside the casing, no groundwater samples were collected from monitoring well MW27 during the October 2003 sampling event. In addition, discrete pre- and post-purging bottom groundwater samples were collected from monitoring well MW26C for analysis of TCL VOCs. To document the sample handling and equipment decontamination procedures, fourteen QA/QC samples were collected during the October 2003 sampling event. These QA/QC samples included three duplicate groundwater samples, two field blank samples, two rinsate samples, three MS samples, three MSD samples, and one effluent sample collected from the carbon adsorber. During transit to the testing laboratory, trip blank samples were placed in each cooler that contained samples for VOC analysis. A total of seven trip blank samples were used during the October 2003 sampling event for documenting the sample handling procedures.

The groundwater samples collected during the October 2003 sampling event were submitted to Lionville Laboratory, Inc. for analysis of TCL VOCs, total concentrations of TAL metals, bicarbonate alkalinity, carbonate alkalinity, chloride, sulfate, sulfide, fluoride, methane, ethane, ethene, nitrate, nitrite, total nitrogen, total dissolved solids, and total organic carbon. At the request of the PADEP, a single effluent sample was collected from the 200-pound carbon adsorber at the end of the October 2003 groundwater sampling event. This treated effluent water sample was submitted to the testing laboratory for analysis of TCL VOCs.

The October 2003 groundwater samples submitted to Lionville Laboratory, Inc. for TCL VOC analysis were collected in 40 mL clear glass VOC vials fixed with hydrochloric acid (HCl) and secured with Teflon®-lined lids. The groundwater samples submitted for the analysis of total TAL metals, cyanide, nitrate, and water quality parameters were collected in 1 liter plastic containers fixed with HNO_3, 1 liter plastic containers fixed with NaOH, 250 mL plastic containers fixed with sulfuric acid (H_2SO_4), and 1 liter plastic containers with no preservatives, respectively. To preserve the integrity of the groundwater samples, the sample containers were stored in coolers at 4°C until delivered to the testing laboratory for analysis. A summary of the analytical methods used to analyze the groundwater samples collected from the monitoring wells during the October 2003 sampling event is outlined in Table 3.3.

During the February 2004 sampling event, groundwater samples were collected from a total of 33 wells (including all monitoring wells and the well at 30 Conestoga Road). Due to an unknown obstruction inside the casing, no groundwater samples were collected from monitoring well MW27 during the February 2004 sampling event. In addition, discrete bottom groundwater samples were collected from 14

of the monitoring wells and the well located at 30 Conestoga Road for analysis of TCL VOCs. To document the sample handling and equipment decontamination procedures, fourteen QA/QC samples were collected during the February 2004 sampling event. These QA/QC samples included two duplicate groundwater samples, two field blank samples, two rinsate samples, two MS samples, two MSD samples, and one effluent sample collected from the carbon adsorber. During transit to the testing laboratory, trip blank samples were placed in each cooler that contained samples for VOC analysis. A total of seven trip blank samples were used during the February 2004 sampling event for documenting the sample handling procedures.

The groundwater samples collected during the February 2004 sampling event were submitted to Lionville Laboratory, Inc. for analysis of TCL VOCs, total and dissolved concentrations of TAL metals, bicarbonate alkalinity, carbonate alkalinity, chloride, sulfate, sulfide, fluoride, methane, ethane, ethene, nitrate, nitrite, total nitrogen, total dissolved solids, and total organic carbon. At the request of the PADEP, a single effluent sample was collected from the 200-pound carbon adsorber at the end of the February 2004 groundwater sampling event. This treated effluent water sample was submitted to the testing laboratory for analysis of TCL VOCs.

The February 2004 groundwater samples submitted to Lionville Laboratory, Inc. for TCL VOC analysis were collected in 40 mL clear glass VOC vials fixed with hydrochloric acid (HCl) and secured with Teflon®-lined lids. The groundwater samples submitted for the analysis of total TAL metals, cyanide, nitrate, and water quality parameters were collected in 1 liter plastic containers fixed with HNO_3, 1 liter plastic containers fixed with NaOH, 250 mL plastic containers fixed with sulfuric acid (H_2SO_4), and 1 liter plastic containers with no preservatives, respectively. To preserve the integrity of the groundwater samples, the sample containers were stored in coolers at 4°C until delivered to the testing laboratory for analysis. A summary of the analytical methods used to analyze the groundwater samples collected from the monitoring wells during the February 2004 sampling event is outlined in Table 3.4.

3.0 GROUNDWATER CHARACTERIZATION RESULTS

The results of the Supplemental Groundwater Characterization of the fractured bedrock aquifer underlying the Bishop Tube site are presented in the following sections.

3.1 Site Stratigraphy

A description of the physical and lithological characteristics exhibited by the rock materials underlying the Bishop Tube site is provided in the following sections. The reader is referred to the Supplemental Soil Characterization Report (Baker, 2003) for more detailed information regarding the soils and weathered bedrock materials underlying the Bishop Tube site.

3.1.1 Bedrock Lithology

Rock cuttings collected during the drilling of the boreholes for monitoring wells MW25, MW26, MW27, and MW28 showed that the bedrock underlying the Bishop Tube site consists of olive gray (2.5Y 5/2) calcareous phyllite and schist with subordinate amounts of calcite and quartz. The carbonate, phyllite, and schist rock fragments recovered from the boreholes of the monitoring wells were found to exhibit characteristics consistent with information published by Sloto (1987) and Senior and others (1997) regarding the Conestoga formation in eastern Chester County, Pennsylvania. It should be noted that the

rocks underlying the Bishop Tube property contained relatively little carbonate material. The rocks underlying the manufacturing complex consisted mainly of non-calcareous phyllite and schist with widely separated thin quartz veins and thin impure calcareous phyllitic layers. The amount of carbonate material contained within the rocks was observed to increase toward the northern edge of the Bishop Tube property. The rock materials generated during the drilling of the borehole for monitoring well MW28 (drilled north of the Norfolk Southern railroad tracks) were found to contain a large amount of calcareous material. The rocks at this location consisted mainly of calcareous phyllites and schist. These observations are consistent with information provided to Baker by local well drillers, indicating that the amount of carbonate material comprising the Conestoga formation dramatically increases north of the Norfolk Southern railroad tracks.

Five water-bearing fracture zones were encountered during the drilling of the borehole for monitoring well MW25. These fractures were encountered at depths of 111 feet, 160 feet, 212 to 222 feet, 250 to 260 feet, and 280 feet, and yielded groundwater at estimated flow rates of 6 gallons per minute (gpm), 6 gpm, 6 gpm, <1 gpm, and 12 gpm, respectively. During the drilling of the borehole for monitoring well MW26, six water-bearing fracture zones were encountered at depths of 20 feet, 31.5 feet, 95 feet, 165 feet, 177 to 187 feet, and 222 feet. The estimated groundwater yields from the water-bearing fracture zones encountered in the borehole for monitoring well MW26 were estimated at 3 gpm, 10 gpm, <1 gpm, 7 gpm, 7 gpm, and 10 gpm, respectively. Six water-bearing fracture zones were encountered during the drilling of the borehole for monitoring well MW27. These fractures were encountered at depths of 57 feet, 67 to 67.5 feet, 90 feet, 175 feet, 223 feet, and 230 feet, and yielded groundwater at estimated flow rates of +40 gpm, 10 gpm, 8 gpm, 10 gpm, <1 gpm, and 1 gpm. During the drilling of the borehole for monitoring well MW28, six water-bearing fracture zones were encountered at depths of 19 feet, 48 to 52 feet, 107 feet, 180 feet, 206 feet, and 227 feet. The estimated groundwater yields from these water-bearing zones in the borehole for monitoring well MW28 were estimated at 3 gpm, 10 gpm, 10 gpm, <1 gpm, <2 gpm, and 6.7 gpm, respectively. In general, the number of water-bearing fractures and the relative magnitude of groundwater yields were found to decrease with respect to depth within the bedrock underlying the Bishop Tube site. This finding is consistent with information published by Sloto (1990) regarding the hydraulic characteristics of the rocks in Chester Valley. The relative elevations of the fractures encountered in the boreholes for monitoring wells MW25, MW26, MW27, and MW28 are shown on the drilling logs presented in Appendix C. The construction of the monitoring wells at the Bishop Tube site and the depths of the water-bearing intervals encountered in the boreholes of the monitoring wells are summarized in Tables 2.1 and 2.2.

It should be noted that during the drilling procedures, difficulties were encountered with the alignment of the boreholes for monitoring wells MW25, MW26, MW27, and MW28. Specifically, the borehole for monitoring well MW25 began to deviate out of alignment/plumbness at a depth of 163 feet. The deviation in the borehole of monitoring well MW25 continued to a depth of 280 feet (i.e., bottom of the well). During the drilling procedures, deviations were also observed between the following intervals in the boreholes for monitoring wells MW26, MW27, and MW28: 105 feet to 250 feet (MW26); 93 feet to 250 feet (MW27); and 125 feet to 250 feet (MW28). The deviation of the boreholes created problems with both stuck and broken drill pipe, required fishing operations to be performed, and extended the drilling program beyond the original planned completion date. In addition, the deviations also interfered with the measurement of the diameter of boreholes (i.e., caliper survey) for monitoring wells MW25, MW26, MW27, and MW28 during the geophysical well logging procedures. The cause of the deviation in the boreholes for monitoring wells MW25, MW26, MW27, and MW28 is unknown, but may be related to the lithology of rock materials encountered in the subsurface (i.e., interbedded phyllitic schists, quartz veins,

and carbonate-rich intervals), differences in hardness exhibited by the interbedded rock materials, and/or the naturally steep dip (approximately 70 to 80 degrees) of the rock foliations.

3.1.2   Geophysical Well Logging Results

As part of the Groundwater Investigation, the open bedrock sections of monitoring wells MW25, MW26, MW27, and MW28 were logged using geophysical techniques to determine the depth of the water-bearing fractures, relative variations in water quality, and changes in lithology. The data provided by the geophysical well logging procedures was used to select intervals for packer testing during the well drilling procedures, as well as to choose the depths of the multi-screened intervals for monitoring wells MW25, MW26, MW27, and MW28. Geophysical well logging techniques were also used to log the well at 30 Conestoga Road to determine the construction of the well, the depth of the water-bearing fractures, relative variations in water quality, and changes in bedrock lithology with respect to depth. The well logging measurements recorded in monitoring wells MW25, MW26, MW27, and MW28, and the well at 30 Conestoga Road are summarized on the geophysical well logs included in Appendix D.

3.1.2.1 Discussion of Log Responses Recorded in Monitoring Well MW25

Monitoring well MW25 was drilled north of Building #8 adjacent to the former vapor degreaser area. The rock materials observed during the drilling procedures consisted mainly of phyllite and schist, with some thin impure carbonate layers and quartz veins. The natural gamma measurements show that the rock materials exposed along the open borehole section of monitoring well MW25 exhibit wide fluctuations in the amount of naturally-occurring radioactivity over short changes in depth (see log plot for MW25 presented in Appendix D). These fluctuations are interpreted to be the result of thin alternating beds of phyllite (high gamma responses) and carbonate-quartz intervals (low gamma responses). The single point resistance, short normal, and long normal responses indicate that the rock materials exposed along the open borehole section of monitoring well MW25 exhibit relatively high resistivity values. A comparison of the natural gamma and resistivity (i.e., single point resistance, short normal, and long normal) responses indicates that in general there is an inverse correlation between these curves. Specifically, intervals exhibiting relatively high gamma responses correlate to low resistivity readings, whereas intervals exhibiting relatively low gamma responses correlate to high resistivity readings (see log plot for MW25 presented in Appendix D). In general, rocks comprised of clay and mica materials typically contain larger amounts of naturally occurring radioactive minerals than carbonates and sandstones, producing higher gamma responses. Carbonate rocks in general exhibit higher responses of resistivity than shales. Based on these relationships, the intervals exhibiting high natural gamma and low resistivity readings in the borehole of monitoring well MW25 are interpreted to represent phyllite/clay-rich intervals, whereas the intervals exhibiting low natural gamma and high resistivity readings are believed to contain a greater proportion of carbonate-quartz minerals.

The highest levels of resistivity were recorded in the borehole of monitoring well MW25 between 230 feet and 250 feet below the ground surface. The spontaneous potential readings recorded within this depth interval show a corresponding low. This information suggests that the rock beds within the interval between 230 and 250 feet are relatively unfractured. It should be noted that yield of groundwater measured during the packer test performed within the interval between 250 and 260 was less than 1 gpm (see Table 1).

The caliper survey suggests that the borehole of monitoring well MW25 encountered fractures within the rock sequence at depths of 111 and 158 feet (see log plot for MW25 presented in Appendix D). Fluctuations of the fluid conductivity and temperature readings within the groundwater below the fractures at 111 and 158 feet collectively suggest that these intervals were yielding water to the borehole. This interpretation is further supported by the positive spontaneous potential response recorded in the borehole just below each of these fractured intervals, suggesting that fresh water is contained within the host rock. The caliper log shows that below a depth of 100 feet, the diameter of the borehole of MW25 steadily decreases. This decrease in diameter is attributed to a deviation of the borehole with respect to depth. The caliper log, resistivity, and spontaneous potential responses suggested that the borehole of monitoring well MW25 encountered water-bearing fractures at the following depths 205 feet, 216 feet, 255 feet, and 275 feet. Based upon the caliper log responses, the resistivity lows, and spontaneous potential highs, the following intervals were selected for packer testing: 106 to 116, 155 to 165, 200 to 210, 212 to 222, 250 to 260, and 272 to 282.

3.1.2.2 Discussion of Log Responses Recorded in Monitoring Well MW26

Monitoring well MW26 was drilled near the northeast corner of the Bishop Tube property. The rock materials observed during the drilling procedures consisted mainly of phyllite and schist, with some thin impure carbonate layers and quartz veins. The natural gamma measurements show that the rock materials exposed along the open borehole section of monitoring well MW26 also exhibit wide fluctuations in the amount of naturally-occurring radioactivity over short changes in depth (see log plot for MW26 presented in Appendix D). These fluctuations are interpreted to be the result of thin alternating beds of phyllite (high gamma responses) and carbonate-quartz intervals (low gamma responses). The single point resistance, short normal, and long normal responses indicate that the rock materials exposed along the open borehole section of monitoring well MW26 exhibit relatively high resistivity values. A comparison of the natural gamma and resistivity (i.e., single point resistance, short normal, and long normal) responses indicates that in general there is an inverse correlation between these curves. Specifically, intervals exhibiting relatively high gamma responses correlate to low resistivity readings, whereas intervals exhibiting relatively low gamma responses correlate to high resistivity readings (see log plot for MW26 presented in Appendix D). In general, rocks comprised of clay and mica materials typically contain larger amounts of naturally occurring radioactive minerals than carbonates and sandstones, producing higher gamma responses. Carbonate rocks in general exhibit higher responses of resistivity than shales. Based on these relationships, the intervals exhibiting high natural gamma and low resistivity readings in the borehole of monitoring well MW26 are interpreted to represent phyllite/clay-rich intervals, whereas the intervals exhibiting low natural gamma and high resistivity readings are believed to contain a greater proportion of carbonate-quartz minerals.

The highest levels of resistivity were recorded in the borehole of monitoring well MW26 between 122 feet and 140 feet below the ground surface. The spontaneous potential readings recorded within this depth interval show a corresponding low. This information suggests that the rock beds within the interval between 122 and 140 feet are relatively unfractured. It should be noted that no yield of groundwater was obtained during the packer test performed within the interval between 118 and 128 feet (see Table 1).

Monitoring well MW26 was drilled east of monitoring well MW25 along a line roughly parallel to the strike of the rock beds in the area. It should be noted that the geophysical log responses for the upper 100 feet of the boreholes for both monitoring wells are roughly similar. This information suggests that the upper portions of the boreholes of monitoring wells MW25 and MW26 may penetrate similar geologic

materials. Below a depth of 100 feet, the geophysical well logs show different responses. These differences are interpreted to be a result of either: 1) facies/lithologic changes, and/or 2) faulting that has disrupted the stratigraphic sequence.

The caliper survey suggests that the borehole of monitoring well MW26 encountered fractures within the rock sequence at depths of 95 feet, 163 feet, 180 feet, and 218 feet (see log plot for MW26 presented in Appendix D). Fluctuations of the fluid conductivity and temperature readings within the groundwater below the fractures at 95, 163, 180, and 218 feet collectively suggest that these intervals were yielding water to the borehole. This interpretation is further supported by the positive spontaneous potential response recorded in the borehole just below each of these fractured intervals, suggesting that fresh water is contained within the host rock. The caliper log shows that below a depth of 105 feet, the diameter of the borehole of MW26 steadily decreases. This decrease in diameter is attributed to a deviation of the borehole with respect to depth. Based upon the caliper log responses, the resistivity lows, and spontaneous potential highs, the following intervals were selected for packer testing: 32 to 37 feet, 45 to 55 feet, 60 to 70 feet, 90 to 100 feet, 118 to 128 feet, 160 to 170 feet, 176 to 186 feet, 195 to 205 feet, and 217 to 227 feet.

3.1.2.3 Discussion of Log Responses Recorded in Monitoring Well MW27

Monitoring well MW27 was drilled along the southern side of Building #8 within the former drum storage area. The rock materials observed during the drilling procedures consisted mainly of phyllite and schist, with minor thin impure carbonate layers and quartz veins. The natural gamma measurements show that the rock materials exposed along the open borehole section of monitoring well MW27 also exhibit wide fluctuations in the amount of naturally-occurring radioactivity over short changes in depth (see log plot for MW27 presented in Appendix D). These fluctuations are interpreted to be the result of thin alternating beds of phyllite/schist (high gamma responses) and carbonate-quartz intervals (low gamma responses). The single point resistance, short normal, and long normal responses indicate that the rock materials exposed along the open borehole section of monitoring well MW27 exhibit relatively high resistivity values. A comparison of the natural gamma and resistivity (i.e., single point resistance, short normal, and long normal) responses indicates that in general there is an inverse correlation between these curves. Specifically, intervals exhibiting relatively high gamma responses correlate to low resistivity readings, whereas intervals exhibiting relatively low gamma responses correlate to high resistivity readings (see log plot for MW27 presented in Appendix D). In general, rocks comprised of clay and mica materials typically contain larger amounts of naturally occurring radioactive minerals than carbonates and sandstones, producing higher gamma responses. Carbonate rocks in general exhibit higher responses of resistivity than shales. Based on these relationships, the intervals exhibiting high natural gamma and low resistivity readings in the borehole of monitoring well MW27 are interpreted to represent phyllite/clay-rich intervals, whereas the intervals exhibiting low natural gamma and high resistivity readings are believed to contain a greater proportion of carbonate-quartz minerals.

Below a depth of 100 feet, the highest levels of resistivity were recorded in the borehole of monitoring well MW27 between 200 feet and 215 feet as measured from the ground surface. The spontaneous potential readings recorded within this depth interval show a corresponding low. This information suggests that the rock beds within the interval between 200 and 215 feet are relatively unfractured. It should be noted that no yield of groundwater was obtained during the packer tests performed within the intervals between 193 to 203 feet and 203 to 213 feet (see Table 1).

The caliper survey suggests that the borehole of monitoring well MW27 encountered fractures within the rock sequence at depths of 43 to 46 feet, 51 to 55 feet, 63 feet, 89 feet, 182 feet, 197 feet, 209 feet, 218 feet, and 222 feet (see log plot for MW27 presented in Appendix D). Fluctuations of the fluid conductivity and temperature readings within the groundwater below the fractures at 43 to 46 feet, 51 to 55 feet, 63 feet, and 182 feet collectively suggest that these intervals were yielding water to the borehole. This interpretation is further supported by the positive spontaneous potential response recorded in the borehole just below each of these fractured intervals, suggesting that fresh water is contained within the host rock. The caliper log shows that below a depth of 95 feet, the diameter of the borehole of MW27 steadily decreases. This decrease in diameter is attributed to a deviation of the borehole with respect to depth. Based upon the caliper log responses, the resistivity lows, and spontaneous potential highs, the following intervals were selected for packer testing: 44 to 65 feet, 50 to 60 feet, 60 to 70 feet, 87 to 97 feet, 126 to 136 feet, 177 to 187 feet, 193 to 203 feet, 203 to 213 feet, and 218 to 228 feet.

3.1.2.4 Discussion of Log Responses Recorded in Monitoring Well MW28

Monitoring well MW28 was drilled offsite along the northern side of the Norfolk Southern railroad tracks. The rock materials observed during the drilling procedures consisted mainly of calcareous phyllite and schist materials. The natural gamma measurements show that the rock materials exposed along the open borehole section of monitoring well MW28 also exhibit wide fluctuations in the amount of naturally-occurring radioactivity over short changes in depth (see log plot for MW28 presented in Appendix D). These fluctuations are interpreted to be the result of thin alternating beds of phyllite/schist (high gamma responses) and carbonate-quartz intervals (low gamma responses). The single point resistance, short normal, and long normal responses indicate that the rock materials exposed along the open borehole section of monitoring well MW28 exhibit relatively high resistivity values. A comparison of the natural gamma and resistivity (i.e., single point resistance, short normal, and long normal) responses indicates that in general there is an inverse correlation between these curves. Specifically, intervals exhibiting relatively high gamma responses correlate to low resistivity readings, whereas intervals exhibiting relatively low gamma responses correlate to high resistivity readings (see log plot for MW28 presented in Appendix D). In general, rocks comprised of clay and mica materials typically contain larger amounts of naturally occurring radioactive minerals than carbonates and sandstones, producing higher gamma responses. Carbonate rocks in general exhibit higher responses of resistivity than shales. Based on these relationships, the intervals exhibiting high natural gamma and low resistivity readings in the borehole of monitoring well MW28 are interpreted to represent phyllite/clay-rich intervals, whereas the intervals exhibiting low natural gamma and high resistivity readings are believed to contain a greater proportion of carbonate-quartz minerals.

The highest levels of resistivity were recorded in the borehole of monitoring well MW28 between 60 feet and 90 feet as measured from the ground surface. The spontaneous potential readings recorded within this depth interval show a corresponding low. This information suggests that the rock beds within the interval between 60 and 90 feet are relatively unfractured. Based upon the geophysical well log responses no intervals were selected for packer testing between 60 and 90 feet.

The caliper survey suggests that the borehole of monitoring well MW28 encountered fractures within the rock sequence at depths of 55 feet, 75 feet, 103 feet, 175 feet, 205 feet, and 220 feet (see log plot for MW28 presented in Appendix D). Fluctuations of the fluid conductivity and temperature readings within the groundwater below the fractures at 55 feet, 103 feet, 175 feet, 205 feet, and 220 feet collectively suggest that these intervals were yielding water to the borehole. This interpretation is further supported by

the positive spontaneous potential response recorded in the borehole just below each of these fractured intervals, suggesting that fresh water is contained within the host rock. The caliper log shows that below a depth of 125 feet, the diameter of the borehole of MW28 steadily decreases. This decrease in diameter is attributed to a deviation of the borehole with respect to depth. Based upon the caliper log responses, the resistivity lows, and spontaneous potential highs, the following intervals were selected for packer testing: 52 to 62 feet, 102 to 112 feet, 118 to 128 feet, 140 to 150 feet, 173 to 183 feet, 201 to 211 feet, and 222 to 232 feet.

3.1.2.5 Discussion of Log Responses Recorded in the Well at 30 Conestoga Road

The well at 30 Conestoga Road is located approximately 1,600 feet northeast of the Bishop Tube site. The well was drilled in September 2002 by Thomas G. Keyes, Inc. of Frazer, Pennsylvania. Based upon a report prepared by Applied Environmental Management, Inc., the well at 30 Conestoga Road was completed as a six-inch diameter open bedrock well. The well was reportedly constructed using 160 feet of six-inch diameter steel casing to seal off the upper interval of the well, and was drilled to a depth of 202 feet. At completion, the well reportedly exhibited a yield of 100 gpm. Groundwater samples collected by Applied Environmental Management, Inc. shortly after the well at 30 Conestoga Road was drilled were found to contain TCE at a concentration of 10,000 μg/l. Other chlorinated solvents detected in the groundwater sample included: 1,1-DCA (190 μg/l), 1,1-DCE (370 μg/l), cis-1,2-DCE (140 μg/l), and 1,1,1-TCA (1,500 μg/l). Based upon the elevated concentrations of chlorinated solvents contained in the groundwater, the well at 30 Conestoga Road is currently not being used for a source of potable water. Geophysical well logging techniques were used to obtain further information regarding the construction of the well, the depths of water-bearing fractures, and the characteristics of the rock materials at this location.

Earth Data Northeast, Inc., used the following geophysical techniques to survey the well at 30 Conestoga Road: natural gamma ray, caliper, spontaneous potential, single point resistance, short normal, and long normal resistivity logs. The well logging work was performed on March 7, 2003.

The geophysical logging instruments indicate that the total depth of the well at 30 Conestoga Road is 163 feet. This information differs from the 202-foot total depth of the well reported on the drilling log prepared by the well driller (see the log plot for the well at 30 Conestoga Road presented in Appendix D). The reason for the disparity between the reported and actual depth of the well at 30 Conestoga Road is unknown, but may be related to the effects of collapse within the open borehole section. It should be noted that the borehole log prepared by the well driller states that large voids were encountered in the subsurface at 162 feet, 170 to 175 feet, and 180 to 185 feet.

The geophysical well log responses indicate that the steel surface casing of the well at 30 Conestoga Road extends to a depth of 158 feet. Accordingly, only a 5-foot interval of open bedrock exists between the bottom of the surface casing and the point of collapse. Based upon this finding, temperature and fluid conductivity logs were not run in the well. The caliper log shows that a relatively large fracture occurs at the bottom of the surface casing. The relatively low gamma ray readings recorded with the open borehole section suggest that the bedrock below the surface casing may be comprised of carbonate materials. This interpretation is consistent with Baker's discussions with other local well drillers, indicating that the Conestoga formation in this area is comprised predominantly of carbonate-rich materials.

3.1.3 FLUTe® NAPL Liner Testing Results

Oleophilic reactive FLUTe® NAPL liners were deployed in the annular space of monitoring wells MW25 and MW27 to evaluate the presence/absence of DNAPLs occurring within the overburden materials (i.e., soils) and fractures exposed along the edges of the boreholes of these wells.

To evaluate the presence/absence and the potential depth of DNAPLs occurring within the overburden materials, a FLUTe® NAPL liner was deployed in the annular space of monitoring well MW25 to a depth of 18 feet. The FLUTe® NAPL liner was deployed in the overburden section of monitoring well MW25 on December 9, 2002, and was allowed to sit in the ground overnight (i.e., +18 hours). The liner was removed on December 10, 2002. The FLUTe® NAPL liner deployed in the overburden portion of the borehole for monitoring well MW25 did not display any evidence of staining or dispersion of the reactive dye, indicating that separate-phase DNAPLs are not present in the subsurface materials where the NAPL liner was deployed. Importantly, the bottom of the NAPL liner did not display evidence of an accumulation of separate-phase DNAPLs in the saturated weathered bedrock materials. This information suggests, that a perched free-phase pool of DNAPLs probably does not exist on top of the bedrock surface underlying the drilling location for monitoring well MW25.

Following the drilling of the 6-inch diameter borehole for monitoring well MW25, a FLUTe® NAPL liner was deployed in the open bedrock section of this well to evaluate the presence or absence of separate-phase chlorinated solvents (i.e., DNAPLs) within the fractures of the bedrock. The borehole of monitoring well MW25 was drilled to a depth of 282 feet below the ground surface. The fractured bedrock section of monitoring well MW25 from 18 feet (bottom of temporary casing) to 282 feet was tested using a FLUTe® NAPL liner. The FLUTe® NAPL liner was deployed in the borehole of monitoring well MW25 on February 10, 2003, and was allowed to sit in the ground overnight (i.e., +18 hours). The liner was removed on February 11, 2003. The FLUTe® NAPL liner deployed in the open borehole section of monitoring well MW25 was found to display evidence of staining and dispersion of the reactive dye at a depth of 280 feet. The groundwater samples collected from the deep-screened section of MW25C (272 to 282 feet) during the March/April 2003, July 2003, and October 2003 quarterly groundwater sampling events were found to contain TCE at 49,000 µg/l, 38,000 µg/l, and 45,000 µg/l, respectively. It should be noted that the aqueous solubility limit for TCE is 1,100,000 µg/l (Cohen and Mercer, 1993). The concentration of TCE measured in the quarterly groundwater samples collected from monitoring well MW25C is greater than 1% of the pure-phase aqueous solubility limit for this solvent. As a general rule of thumb, Johnson and Pankow (1992) suggest that dissolved concentrations of TCE in groundwater equal to or greater than 1% of this solvent's pure-phase aqueous solubility limit (i.e., 11,000 µg/l) is an indication that DNAPLs may be present in the subsurface. The results of the FLUTe® NAPL liner testing and the analytical results for the quarterly groundwater samples collectively suggest that residual DNAPLs probably occur in the fractured bedrock materials at a depth of 280 feet in the vicinity of monitoring well MW25.

To evaluate the presence/absence and the potential depth of DNAPLs occurring within the overburden materials, a FLUTe® NAPL liner was deployed in the annular space of monitoring well MW27 to a depth of 45 feet. The FLUTe® NAPL liner was deployed in the overburden section of monitoring well MW27 on December 9, 2002, and was allowed to sit in the ground for +3 hours. The liner was removed on December 9, 2002. The FLUTe® NAPL liner deployed in the overburden portion of the borehole for monitoring well MW27 did not display any evidence of staining or dispersion of the reactive dye, indicating that separate-phase DNAPLs are not present in the subsurface materials where the NAPL liner

was deployed. Importantly, the bottom of the NAPL liner did not display evidence of an accumulation of separate-phase DNAPLs in the weathered bedrock materials. This information suggests, that a perched separate-phase pool of DNAPLs probably does not exist on top of the bedrock surface underlying the drilling location for monitoring well MW27.

Following the drilling of the 6-inch diameter borehole for monitoring well MW27, a FLUTe® NAPL liner was deployed in the open bedrock section of this well to evaluate the presence or absence of separate-phase chlorinated solvents (i.e., DNAPLs) within the fractures of the bedrock. The borehole of monitoring well MW27 was drilled to a depth of 242 feet below the ground surface. The fractured bedrock section of monitoring well MW27 from 45 feet (bottom of temporary casing) to 242 feet was tested using a FLUTe® NAPL liner. The FLUTe® NAPL liner was deployed in the borehole of monitoring well MW27 on December 16, 2002, and was allowed to sit in the ground overnight (i.e., +18 hours). The liner was removed on December 17, 2002. The FLUTe® NAPL liner deployed in the open borehole section of monitoring well MW27 was not found to display evidence of staining and dispersion of the reactive dye. This information suggests that residual DNAPLs probably do not occur in the fractured bedrock materials in the vicinity of monitoring well MW27.

3.1.4   Analytical Results for Groundwater Samples Collected During Packer Testing Activities

To assess changes in the concentrations of organic compounds with respect to depth in the boreholes for monitoring wells MW25, MW26, MW27, and MW28, groundwater samples were collected from selected fracture zones using a straddle-packer apparatus. The groundwater quality data furnished by the packer testing procedures was used in conjunction with other drilling information to design the construction of the multi-screened intervals within the boreholes of monitoring wells MW25, MW26, MW27, and MW28. A 2-inch submersible pump along with a straddle-packer apparatus was utilized to collect the depth-discrete groundwater samples from each monitoring well. To estimate groundwater yields, a flow meter was connected to the pump discharge hose. The packer testing intervals were selected based upon: 1) information collected by the onsite geologist during the drilling procedures; 2) the evaluation of the geophysical well logs; and 3) the FLUTe® NAPL liner testing results (monitoring wells MW25 and MW27 only). A summary of the packer testing intervals and groundwater yields measured in the boreholes of monitoring wells MW25, MW26, MW27, and MW28 are presented in Table 1.

Six intervals were selected for packer testing in the borehole of monitoring well MW25 (see Table 1). Groundwater samples were collected for rapid turn-around (24-hour) analysis from the following five water-bearing intervals: 106 to 116 feet; 155 to 165 feet; 212 to 222 feet; 250 to 260 feet; and 272 to 288 feet. The following chlorinated VOCs were detected in the groundwater samples collected during the packer testing procedures performed in the borehole of monitoring well MW25: 1,1-DCE, cis-1,2-DCE, trans-1,2-DCE, PCE, 1,1,1-TCA, TCE, and vinyl chloride. The concentrations of 1,1-DCE, cis-1,2-DCE, PCE, 1,1,1-TCA, TCE, and vinyl chloride were found to exceed the PADEP Non-Residential Health-based Groundwater Standards in each packer test interval (see Table 7.1). The highest concentrations of VOCs were detected in the groundwater samples collected from the 106 to 116 foot, the 155 to 165 foot, and the 272 to 282 foot intervals. In general, the concentrations of chlorinated VOCs increased with respect to depth in the borehole of monitoring well MW25. The highest groundwater yields (estimated at 6 gpm) were measured within the following borehole intervals of monitoring well MW25: 106 to 116 feet, 155 to 165 feet, 212 to 222 feet, and 272 to 282 feet. Based upon the groundwater quality data, the estimated groundwater yields, the geophysical well logging results, and the FLUTe® NAPL liner testing

results, the following intervals were selected to be screened within the borehole of monitoring well MW25: 106 to 116 feet, 212 to 222 feet, and 272 to 282 feet.

Following the drilling of the 6-inch diameter borehole for monitoring well MW26, nine intervals were selected for packer testing (see Table 1). Groundwater samples were collected for rapid turn-around (24-hour) analysis from the following five water-bearing intervals: 32 to 37 feet; 90 to 100 feet; 160 to 170 feet; 176 to 186 feet; and 217 to 227 feet. The following chlorinated VOCs were detected in the groundwater samples collected during the packer testing procedures performed in the borehole of monitoring well MW26: 1,1-DCE, cis-1,2-DCE, trans-1,2-DCE, PCE, 1,1,1-TCA, TCE, and vinyl chloride. The concentrations of 1,1-DCE, cis-1,2-DCE, PCE, 1,1,1-TCA, TCE, and vinyl chloride were found to exceed the PADEP Non-Residential Health-based Groundwater Standards in the 176 to 186 foot packer test interval (see Table 7.2). The analytical results show that the concentrations of 1,1-DCE, cis-1,2-DCE, PCE, TCE, and vinyl chloride were found to exceed the PADEP Non-Residential Health-based Groundwater Standards in the 32 to 37 foot, the 90 to 100 foot, the 160 to 170 foot, and the 217 to 227 foot intervals. The highest concentrations of VOCs were detected in the groundwater samples collected from the 176 to 186 foot and the 217 to 227 foot intervals. In general, the concentrations of chlorinated VOCs increased to a depth of 186 feet within the borehole of monitoring well MW26. The concentration of TCE measured in the groundwater sample collected from the 176 to 186 foot interval was 176,000 µg/l. It should be noted that the aqueous solubility limit for TCE is 1,100,000 µg/l (Cohen and Mercer, 1993). The concentration of TCE measured in the groundwater sample collected from the 176 to 186 foot packer test interval is greater than 1% of the pure-phase aqueous solubility limit for this solvent. As a general rule of thumb, Johnson and Pankow (1992) suggest that dissolved concentrations of TCE in groundwater equal to or greater than 1% of this solvent's pure-phase aqueous solubility limit (i.e., 11,000 µg/l) is an indication that DNAPLs may be present in the subsurface. Separate-phase DNAPLs were not observed in the groundwater sample collected from the 176 to 186 foot packer test interval. The groundwater sampling data, however, suggest that separate-phase DNAPLs may exist in the fractures of the host bedrock within the vicinity of monitoring well MW26. The highest groundwater yields were measured within the following borehole intervals of monitoring well MW26: 32 to 37 feet (10 gpm), 160 to 170 feet (7 gpm), 176 to 186 feet (7 gpm), and 217 to 227 feet (10 gpm). Based upon the groundwater quality data, the estimated groundwater yields, and the geophysical well logging results, the following intervals were selected to be screened within the borehole of monitoring well MW26: 90 to 100 feet, 176 to 186 feet, and 222 to 232 feet.

Nine intervals were selected for packer testing in the borehole of monitoring well MW27 (see Table 1). Groundwater samples were collected for rapid turn-around (24-hour) analysis from the following six water-bearing intervals: 44 to 65 feet; 50 to 60 feet; 60 to 70 feet; 87 to 97 feet; 177 to 187 feet; and 218 to 228 feet. The following chlorinated VOCs were detected in the groundwater samples collected during the packer testing procedures performed in the borehole of monitoring well MW27: 1,1-DCE, cis-1,2-DCE, trans-1,2-DCE, PCE, 1,1,1-TCA, and TCE. The concentrations of TCE were found to exceed the PADEP Non-Residential Health-based Groundwater Standards in the 50 to 60 foot, 60 to 70 foot, 87 to 97 foot, 177 to 187 foot, and 218 to 228 foot packer test intervals (see Table 7.3). The highest concentrations of VOCs were detected in the groundwater samples collected from the 177 to 187 foot and the 218 to 228 foot intervals. In general, the concentrations of chlorinated VOCs increased with respect to depth in the borehole of monitoring well MW27. The highest groundwater yields were measured within the following borehole intervals of monitoring well MW27: 50 to 60 feet (+10 gpm), 60 to 70 feet (+10 gpm), 87 to 97 feet (8 gpm), and 177 to 187 feet (+10 gpm). Based upon the groundwater quality data, the estimated groundwater yields, and the geophysical well logging results, the following intervals were selected to be

screened within the borehole of monitoring well MW27: 87 to 97 feet, 177 to 187 feet, and 230 to 240 feet.

Following the drilling of the 6-inch diameter borehole for monitoring well MW28, seven intervals were selected for packer testing (see Table 1). Groundwater samples were collected for rapid turn-around (24-hour) analysis from the following five water-bearing intervals: 52 to 62 feet; 102 to 112 feet; 173 to 183 feet; 201 to 211 feet; and 222 to 232 feet. The following chlorinated VOCs were detected in the groundwater samples collected during the packer testing procedures performed in the borehole of monitoring well MW28: 1,1-DCE, cis-1,2-DCE, PCE, 1,1,1-TCA, TCE, and vinyl chloride. The concentrations of 1,1-DCE, cis-1,2-DCE, PCE, 1,1,1-TCA, TCE, and vinyl chloride were found to exceed the PADEP Non-Residential Health-based Groundwater Standards in the 52 to 62 foot; 102 to 112 foot; 173 to 183 foot; 201 to 211 foot; and 222 to 232 foot packer test intervals (see Table 7.4). The highest concentrations of VOCs were detected in the groundwater samples collected from the 201 to 211 foot and the 222 to 232 foot intervals. In general, the concentrations of chlorinated VOCs increased with respect to depth in the borehole of monitoring well MW28. The concentration of TCE measured in the groundwater sample collected from the 222 to 232 foot interval was 15,200 μg/l. It should be noted that the aqueous solubility limit for TCE is 1,100,000 μg/l (Cohen and Mercer, 1993). The concentration of TCE measured in the groundwater sample collected from the 222 to 232 foot packer test interval is greater than 1% of the pure-phase aqueous solubility limit for this solvent. As a general rule of thumb, Johnson and Pankow (1992) suggest that dissolved concentrations of TCE in groundwater equal to or greater than 1% of this solvent's pure-phase aqueous solubility limit (i.e., 11,000 μg/l) is an indication that DNAPLs may be present in the subsurface. Separate-phase DNAPLs were not observed in the groundwater sample collected from the 222 to 232 foot packer test interval. The groundwater sampling data, however, suggest that separate-phase DNAPLs may exist in the fractures of the host bedrock within the vicinity of monitoring well MW28. The highest groundwater yields were measured within the following borehole intervals of monitoring well MW28: 52 to 62 feet (10 gpm), 102 to 112 feet (+10 gpm), and 222 to 232 feet (+7 gpm). Based upon the groundwater quality data, the estimated groundwater yields, and the geophysical well logging results, the following intervals were selected to be screened within the borehole of monitoring well MW28: 100 to 110 feet, 173 to 183 feet, and 222 to 232 feet.

3.1.5   Analytical Results for the Treated Water Samples Collected from the Carbon Adsorber

The groundwater generated during the drilling of the boreholes for monitoring wells MW25, MW26, MW27, and MW28, was captured and transferred to an 18,000-gallon capacity frac tank for temporary storage. Periodically during the drilling work, Baker pumped the water contained in the frac tank through a 1,000-pound liquid-phase carbon adsorber. The treated water was ultimately discharged to the ground surface at the site. To evaluate the effectiveness of the carbon adsorber in removing the VOCs contained in the raw groundwater, samples of the treated effluent were collected at the end of each treatment event. The treated effluent samples were submitted to Analytical Laboratory Services, Inc. for overnight (i.e., 24-hour turnaround) analysis of selected chlorinated VOCs. The analytical results for the treated effluent samples collected from the carbon adsorber are summarized in Tables 5.1 and 5.2.

The testing results show that the concentrations of the selected chlorinated VOC parameters were all below the PADEP Non-Residential Statewide Health-based Groundwater Standards. Trace concentrations of TCE (i.e., "J" values) were detected in the treated groundwater samples collected on January 15, February 3, and March 20, 2003 (see Table 5.1). Importantly, the analytical results indicate that the treatment and discharge of the groundwater generated during the drilling of the monitoring wells was

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performed in accordance with the pre-approved testing and handling procedures stipulated by the PADEP Bureau of Water Quality.

3.2   Conceptual Geologic/Hydrogeologic Model

The Valley Creek Basin is situated within Chester Valley. This area is included within the Piedmont Physiographic Province (see Figure 3). The Valley Creek Basin contains carbonate-rich rocks (limestone and dolomite) bounded to the north and south by non-carbonate rocks (mostly metamorphic phyllites and schists). Geologically, the Chester Valley is underlain by the Peach Bottom synform, which trends about N72ºE. The carbonate rocks dip very steeply (80º from the horizontal) within the synform and underlie almost 70% of the land area in Chester Valley. The remaining area is underlain by non-carbonate rocks, which comprise the walls of the valley due to their greater resistance to weathering and erosion (Crawford et al., 1999). A more detailed description of the regional geology is presented in the Phase I Site Characterization report (Baker, 2002a).

The formations of interest to the Groundwater Investigation and the development of the groundwater flow model range in age from Pre-Cambrian to Ordovician. The Bishop Tube site is underlain by the Conestoga formation (O€c) and is situated near the southern contact with the Octoraro Phyllite (€Zo, or Xwc in older reports) (see Figure 4). The Conestoga Limestone is comprised of gray, finely crystalline limestone, argillaceous limestone with some graphitic to micaceous limestone, and some angular clasts in a calcareous matrix. The Conestoga formation reportedly also includes crystalline, sandy or silty limestone and dark-gray dolomite (Crawford et al., 1999). At the Bishop Tube site the Conestoga formation is mainly comprised of phyllite and schist with thin interbeds of impure limestone.

The Octoraro Phyllite is a localized name for the greenschist-facies phyllite of the Octoraro Formation and is thought to be a low-grade metamorphic equivalent of the much more extensive Wissahickon formation (Crawford et al., 1999). The Octoraro Phyllite is a non-carbonate rock. Groundwater flow in the formation is principally through secondary fractures. The formation is very resistant to weathering and forms the steep hillside south of the Bishop Tube site, exhibiting a local change in relief of about 250 feet within one-half mile (~10% slope).

Groundwater occurs within both the overburden and in the rocks underlying the Valley Creek Basin (including the Bishop Tube site). According to Sloto (1990), primary porosity/permeability is virtually non-existent in the carbonate rocks underlying the Chester Valley area. Groundwater contained in the rocks is produced mainly from secondary openings (i.e., fractures, solution cavities, etc.). These fractures are believed to have been produced by various natural stresses/forces superimposed upon the rocks. The number and size of the water-bearing openings determines the amount of secondary porosity contained in the rock. The number, size, and degree of interconnection of the openings influence the magnitude of the secondary permeability. Where the rocks are extensively fractured, permeability may be high, whereas elsewhere in the same unit the rock may be nearly impermeable. The majority of the secondary fractures are oriented such that groundwater moves preferentially parallel to the strike of the valley trend (i.e., toward the northeast). The hydrogeology of Chester Valley is dominated by this characteristic. Based upon this relationship, the hydraulic conductivity (K) value of each hydrogeologic unit is believed to be anisotropic, with the K_x value (measured parallel to bedding) being about five times greater than the permeability measured perpendicular to bedding (K_y)(Sloto, 1990). This information collectively suggests that the rate of groundwater flow (and hence the migration of dissolved solutes) may be greater either in zones where the underlying bedrock is extensively fractured and/or parallel to the strike of the underlying

rock beds. These characteristics are believed to be important in controlling the migration of solutes dissolved in the fractured bedrock aquifer underlying the area in the vicinity of the Bishop Tube site.

In the Chester Valley area, the weathered zone overlying the fractured bedrock interval is particularly significant to the local hydrogeologic system. Sloto (1990) reports that this weathered zone is typically comprised of a mixture of weathered rock, sand, silt, and clay materials, exhibiting a moderate to low permeability. This interval, however, is generally porous and contains considerable amounts of water in storage. Saturated conditions were observed within the weathered bedrock zone in the borings drilled at the Bishop Tube site during the Phase I Site Characterization (Baker, 2002a). Based upon these observations and published information, it is believed that recharge to the fractured bedrock interval underlying the Bishop Tube site and elsewhere in Chester Valley is provided in part from groundwater locally contained in the overlying saturated weathered zone.

Groundwater contained in the weathered bedrock and fractured bedrock intervals of the Chester Valley area was modeled by Sloto (1990) to occur under water table conditions. On a local scale at the Bishop Tube site, the groundwater contained in the weathered bedrock and fractured bedrock intervals may actually occur under semi-confined flow conditions. The water levels measured in nested monitoring well pairs MW02-MW03-MW19 (now abandoned), showed a potentiometric difference between the shallow and deep portions of the underlying fractured bedrock aquifer. Historically, the groundwater contained in monitoring wells MW02 and MW03 (i.e., shallow wells) was over 12 feet higher in elevation than the potentiometric level recorded in monitoring well MW19 (i.e., deep well). This information indicates that shallow groundwater may locally exhibit a higher hydraulic potential than groundwater contained in the deeper portions of the fractured bedrock aquifer underlying the site. The construction of the monitoring wells and a historical record of the groundwater levels recorded at the Bishop Tube site are summarized in Tables 2 and 8, respectively. Importantly, this information suggests that the higher hydraulic potential exhibited by groundwater contained within the shallow portion of the Conestoga formation underlying the Bishop Tube site may be providing an additional pressure gradient for the downward migration of DNAPLs to deeper levels of the fractured bedrock aquifer.

The rock beds underlying the area in the vicinity of the Bishop Tube site exhibit a relatively steep dip (i.e., 80 degrees). The dip of the rock beds perpendicular to strike underlying the Bishop Tube site is displayed on Figure 5. The stratigraphic section parallel to strike is displayed in Figure 6. O'Brien and Gere (1998) reported observing nearly horizontal and bedding parallel fractures in the rock core collected from the borehole of monitoring well MW19 (now abandoned). The nearly horizontal fractures observed by O'Brien and Gere (1998) in the rock core from monitoring well MW19 appeared to be enlarged by solution of the more soluble (carbonate intervals) within the rock. The apertures of the bedding parallel fractures that occur along the phyllites/schist foliations could not be assessed by O'Brien and Gere (1998). Information published by Sloto (1990, 2001) suggests that the apertures of the bedding fractures contained within the Conestoga formation are generally on the order of fractions of an inch. Larger fractures may occur locally near the surface of the bedrock, resulting from weathering and dissolution processes. The steeply dipping fractures/laminations within the stratigraphic sequence were interpreted by O'Brien and Gere (1998) to provide conduits of flow both subvertically and parallel to the direction of strike of the rock beds. It should be noted that although elevated concentrations (i.e., locally greater than 4,000,000 µg/kg) of chlorinated solvents have been identified in the soils underlying the source areas, no "perched" separate-phase layers of DNAPLs have been identified in the overburden materials or on the top of bedrock at the site (Baker, 2002a and 2003). Importantly, this finding suggests that the internal structure and hydraulic conductivity of the soils, weathered bedrock (i.e., saprolite with steeply dipping

relic bedding structure), and phyllite/schist rock materials have allowed any separate-phase DNAPLs to migrate downward and enter the foliations, bedding plane partings, and fractures of the underlying bedrock. Information published by Kueper and others (2003) suggests that separate-phase chlorinated DNAPL pools situated within overburden materials at the top of bedrock are generally unlikely, because the fracture apertures required to prevent DNAPL migration need to be less than 100 microns (μm) in size. For comparison purposes, a human hair exhibits an approximate thickness of 50 μm. This information collectively suggests that that the steep inclination (80^0) and large aperture openings of the foliations and vertical fractures contained in the phyllite and schist materials have collectively allowed the chlorinated solvents released at the Bishop Tube site to penetrate and migrate in a down-dip direction within the bedrock underlying the source areas.

The steep dip of the foliations/bedding planes as depicted on Figure 5 indicates that the boreholes of the deep monitoring wells drilled at the Bishop Tube site (see monitoring wells MW26 and MW27) penetrate only a thin interval of the stratigraphic section. As discussed above, the steeply dipping foliations/bedding planes are believed to serve as conduits for the migration of DNAPLs to deeper levels of the underlying fractured bedrock aquifer. This supposition is supported by the analytical results for the groundwater samples collected from monitoring well MW27B (see Section 3.5). The groundwater samples collected from the screened interval of monitoring well MW27B have been found to contain elevated concentrations of TCE ranging from 11,000 μg/l to 28,000 μg/l. As discussed in Section 3.5, the elevated concentrations of TCE suggest that separate-phase DNAPLs may be present in the vicinity of the screened interval of monitoring well MW27B. Monitoring well MW27B was drilled south of the former drum storage area (see Figure 2). Based upon the direction of groundwater flow, monitoring well MW27B is situated hydraulically upgradient to the former drum storage area.

The cross-section A to A' (see Figure 5) shows that the updip location for the screened section of monitoring well MW27B (depth of 177 to 187 feet) extends back to the ground surface at a point corresponding to the location of the former drum storage area (in the vicinity of monitoring well MW22 on cross-section A to A'). Elevated concentrations of TCE (i.e., +4,000,000 μg/kg) have been locally detected in the soils of the former drum storage area (Baker, 2002). Based upon this information, DNAPLs released in the former drum storage area are believed to have migrated downdip along the steeply dipping foliations/bedding planes to provide the source for the elevated concentrations of TCE detected in the screened interval of monitoring well MW27B. According to Cohen and Mercer (1993), groundwater concentration trends increasing in a hydraulically upgradient direction is an indication that separate-phase DNAPLs may be present in the subsurface. Such trends are generally caused by the preferential migration of DNAPLs along permeable conduits that exhibit slopes counter to the hydraulic gradient.

3.3   Onsite Groundwater Flow

Water level elevations were measured periodically in the monitoring wells during the investigation. The static water levels (SWLs) recorded in the monitoring wells during the study are listed in Tables 8.1 through 8.3. For comparison purposes, the elevation of the top of the surface casing for each of these monitoring points was surveyed to mean sea level. A summary of the construction of the monitoring wells at the Bishop Tube site is displayed in Tables 2.1 and 2.2. The locations of the onsite monitoring wells are shown on Figure 2. The locations of the offsite wells (i.e., 30 Conestoga Road and 54 Conestoga Road) are presented in Figure 7.

As discussed in Section 3.2, the groundwater contained within the Conestoga formation underlying the Bishop Tube property is believed to occur under water table and semi-confined conditions. This supposition is based upon: 1) groundwater elevation differences measured between shallow and deep nested well pairs; and 2) the presence of flowing wells (i.e., artesian effects) at the site. Based upon published information (Sloto, 1990; McManus and Sloto, 1994) the groundwater flow patterns within the Conestoga formation are generally parallel to the regional topographic gradient. Locally, groundwater may discharge as baseflow to the gaining portions of local streams.

Relatively higher water elevations recorded in monitoring well MW07 and MW24, and lower water elevations measured in monitoring well MW16 suggest that groundwater flow in the water table aquifer underlying the Bishop Tube site on March 31, 2003 was toward the north-northeast (see Figure 8). The directions of groundwater flow as depicted on Figure 8 are consistent with the local topographic differences in the vicinity of the Bishop Tube site. In addition, this finding is also consistent with previous studies performed at the site (O'Brien and Gere, 1998; Baker, 2002b) and information published by Sloto (1984, 1987, and 1990) and McManus and Sloto (1994) showing that groundwater flow within the carbonate rocks of the local area is toward the north-northeast. The groundwater elevations measured in the monitoring wells during the July 2003 and October 2003 sampling events showed that groundwater flow within the water table flow system was also toward the north-northeast.

The relatively higher groundwater elevations recorded in monitoring well MW01 and lower water elevations measured in monitoring well MW28A suggest that groundwater flow in the deeper portions of the bedrock aquifer on March 31, 2003 was also toward the north-northeast (see Figure 9). The directions of groundwater flow as depicted on Figure 9 are consistent with the local topographic differences in the vicinity of the Bishop Tube site. In addition, this finding is also consistent with previous studies performed at the site (O'Brien and Gere, 1998; Baker, 2002b) and information published by Sloto (1984, 1987, and 1990) and McManus and Sloto (1994) showing that groundwater flow within the carbonate rocks of the local area is toward the north-northeast. The groundwater elevations measured in the monitoring wells during the July 2003 and October 2003 sampling events showed that groundwater flow within the bedrock flow system was also toward the north-northeast.

The March 31, 2003 water table flow map (Figure 8) indicates that the hydraulic gradient of the water table in the central portion of the Bishop Tube property was 0.047 feet per foot. The hydraulic gradient of the potentiometric surface representing groundwater flow within the deeper portion of the bedrock aquifer (Figure 9) on March 31, 2003 was 0.054 feet/foot.

Based upon a comparison of water table elevations to surface elevations, shallow groundwater appears to be serving as a source of baseflow to Little Valley Creek along the eastern portion of the Bishop Tube property. It should be noted that a surface water study completed by the PADEP in May 2003 (PADEP, 2003) found elevated concentrations of TCE (i.e., 55 µg/l) in the surface water of Little Valley Creek near the northeast corner of the Bishop Tube property. Groundwater samples collected from monitoring well MW08 (situated near the northeast corner of the Bishop Tube property) have been found to contain elevated concentrations of TCE ranging from 260 µg/l in March/April 2003 to 410 µg/l in October 2003. This information is consistent with the finding outlined above that shallow groundwater is serving as source of baseflow for Little Valley Creek along the eastern portion of the Bishop Tube property.

Static water levels measured in nested well pairs MW25 (A, B, and C), MW26 (A, B, and C), and MW27 (A, B, and C) show that groundwater contained in the upper section of the bedrock exhibits a higher

hydraulic potential than the groundwater contained within the deeper portion of the aquifer in the vicinity of these monitoring wells. Groundwater levels recorded in nested well pairs MW15/MW16 and MW28 (A, B, and C), however, show that groundwater contained within the bedrock aquifer is discharging to the water table flow system in the vicinity of these monitoring wells. It should be noted that on March 21 and 31, 2003, groundwater was observed discharging over the top of the casing of monitoring well MW15 (i.e., flowing well). Based upon this information, the groundwater within the bedrock aquifer may also be serving as a supplemental source of baseflow to Little Valley Creek northeast of the Bishop Tube site.

To evaluate hydraulic head differences between the water table and bedrock flow systems underlying the Bishop Tube site, potentiometric cross-sections were constructed (see Figures 10 and 11). These cross-sections display differences in potentiometric head along lines perpendicular and parallel to the strike of the foliations/bedding contained within the underlying phyllite/schist rocks. The cross-section along A to A' (i.e., Figure 10) was prepared using a dip angle of 80 degrees. This dip angle was corrected for a 3:1 vertical exaggeration used to show changes in topographic elevation on the cross-section. It should be noted that the bedding planes depicted on cross-section A to A' are shown for reference purposes only and do not reflect actual bed thicknesses.

To further evaluate groundwater flow within the water table and bedrock flow systems, groundwater contour cross-sections were prepared for the directions perpendicular and parallel to the direction of strike of the underlying rock beds (see Figures 10 and 11, respectively). As shown on Figures 10 and 11, the water table extends above the top of the bedrock. The groundwater contours depicted on Figure 10 indicate that in general groundwater flow in the water table flow system is horizontal. This horizontal flow component for the unconfined flow system is believed to be controlled by the top of bedrock. Along the southern portion of the Bishop Tube site, groundwater in the bedrock exhibits a downward flow component (see Figure 10). This downward flow component is interpreted to be a function of groundwater recharge along the hillside south of the Bishop Tube site. In the area underlying the central and northern portions of the Bishop Tube property, groundwater flow in the bedrock is primarily horizontal (see Figure 10). Based upon the groundwater elevations measured on March 31, 2003, groundwater flow parallel to strike in the water table and bedrock systems exhibits a downward flow component (see Figure 11). Importantly, this downward flow component may be functioning as a supplemental pressure head for the migration of dissolved-phase and separate-phase solutes to deeper levels in the underlying fractured bedrock aquifer. As mentioned above, the hydraulic conductivity of the rocks underlying the area is much greater parallel to the direction of strike than flow perpendicular to the bedding planes. This information regarding the groundwater flow patterns and hydraulic characteristics of the aquifer is believed to be important for the migration of the dissolved-phase and separate-phase solutes from the Bishop Tube site.

As discussed above, Little Valley Creek adjacent to and immediately downgradient of the Bishop Tube Property appears to be receiving baseflow from the shallow water table flow system. It should be noted that elevated concentrations of TCE were detected in the groundwater samples collected from several deep monitoring wells at the Bishop Tube site (see Section 3.5). To more fully evaluate groundwater flow within the bedrock aquifer underlying the area in the vicinity of the Bishop Tube site, a three-dimensional groundwater flow model was developed as part of the Supplemental Groundwater Investigation. The results of the groundwater flow model (see Section 5.3) suggest that groundwater in the deeper levels of the bedrock aquifer underlying the Bishop Tube property may ultimately be serving as a source of baseflow to Little Valley Creek farther northeast of the site. As will be discussed in Section 5.6, the analytical results for surface water samples collected by the PADEP in May 2003 from Little Valley

Creek show an increase in the concentration of TCE northeast of the Bishop Tube site. This increase in the concentration of TCE is speculated to be the result of deep groundwater originating from the Bishop Tube site discharging to Little Valley Creek. These flow patterns and findings are important because Little Valley Creek has been classified by the PADEP (Pennsylvania Code, Title 25, Environmental Resources, Chapter 93 – *Water Quality Standards)* as an "*Exceptional Value*" stream. The presence of TCE within the surface water of Little Valley Creek may pose an ecological risk to the indigenous flora and fauna.

3.4   Groundwater Samples Collected From Residential Wells

Baker collected groundwater samples from the private water supply well located at 54 Conestoga Road during the March/April 2003 and the October 2003 sampling events. The groundwater samples were collected before the treatment system, between the carbon filters, and after the treatment system. The groundwater samples collected from the private water supply well at 54 Conestoga Road were sent to Severn Trent Laboratories, Inc. (March/April 2003 event) and Lionville Laboratory, Inc. (October 2003 event) for the analysis of TCL VOCs. The pre-filter samples collected in October 2003 were also analyzed for total metals and water quality parameters. The testing results for the groundwater samples collected from the private water supply well at 54 Conestoga Road during the investigation are summarized on Table 9.8 (March/April 2003 event), and Tables 18.10 and 19.9 (October 2003 event).

Concentrations of total 1,2-DCE (5 μg/l) and 1,1,1-TCA (2 μg/l) were detected below the PADEP Residential Statewide Health-based Groundwater Standards in the pre-treatment groundwater sample collected from the residential well during the March/April 2003 sampling event. In addition, TCE was detected at 19 μg/l in the pre-treatment groundwater sample collected from the residential well. The concentration of TCE detected in the sample exceeded the PADEP Residential Statewide Health-based Groundwater Standard of 5 μg/l established for this parameter. No VOCs were detected in the mid- or post-treatment groundwater samples.

The analytical results for the groundwater samples collected from the residential well during the October 2003 sampling event showed that concentrations of 1,1-DCA, total 1,2-DCE, 1,1,1-TCA, and TCE were detected in the pre-treatment sample. The concentrations of 1,1-DCA, total 1,2-DCE, and 1,1,1-TCA were all detected below the PADEP Residential Statewide Health-based Groundwater Standards established for these parameters. The concentration of TCE, however, was measured at 24 μg/l, exceeding the PADEP Residential Statewide Health-based Groundwater Standard of 5 μg/l. No VOCs were detected in the mid- or post-treatment samples. Accordingly, the carbon treatment system at 54 Conestoga Road appears to be properly lowering the VOC concentrations dissolved in the groundwater being supplied by the private well.

The pre-treatment sample collected from the residential well located at 54 Conestoga Road during the October 2003 sampling event was also analyzed for total metals. The following heavy metals were detected in the pre-treated groundwater sample collected in October 2003: antimony (1.3 μg/l), barium (35 μg/l), chromium (2.6 μg/l), copper (6.8 μg/l), nickel (2.4 μg/l), selenium (0.75 μg/l), and zinc (5.8 μg/l). The concentrations of these heavy metals measured in the pre-treated groundwater sample were all below the PADEP Residential Statewide Health-based Groundwater Standards established for these parameters. Lead was detected in the pre-treatment groundwater sample at a concentration of 7.9 μg/l. The measured concentration of lead in the pre-treatment groundwater sample exceeded the PADEP Residential Statewide Health-based Groundwater Standard of 5 μg/l. In addition, low-levels of fluoride

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(0.82 mg/l) and nitrate (0.85 mg/l) were detected in the pre-treatment groundwater sample collected from the residential well located at 54 Conestoga Road. The source of the elevated concentrations of lead in the raw groundwater being supplied to the home at 54 Conestoga Road is unknown. It should be noted that the home at 54 Conestoga Road is believed to have been constructed in the early 1950's. Based upon this information, the elevated concentrations of lead may be related to the type of piping materials (i.e., copper piping with lead solder) used to connect the well to the internal plumbing system of the house.

3.5   Groundwater Samples Collected From Monitoring Wells

Four rounds of groundwater samples were collected from the 33 monitoring wells and the well located at 30 Conestoga Road during the following time periods: March 31 through April 8, 2003 (first round), July 14 through July 23, 2003 (second round), October 20 through October 28, 2003 (third round), and February 9 through February 17, 2004 (fourth round). The private water supply well located at 54 Conestoga Road was sampled during the March/April 2003 and the October 2003 groundwater sampling events. The testing results for the first round of groundwater samples collected from the monitoring wells, the well at 30 Conestoga Road, and the private well located at 54 Conestoga Road are summarized in Tables 9.1 through 9.10, Table 10, and Tables 11.1 through 11.8. The second round testing results are summarized in Tables 12.1 through 12.9, Tables 13.1 through 13.2, Tables 14.1 through 14.8, Tables 15.1 through 15.8, Tables 16.1 through 16.5, and Tables 17.1 through 17.2. The testing results for the third round of groundwater samples collected from the 33 monitoring wells, the well at 30 Conestoga Road, and the private well at 54 Conestoga Road are summarized in Tables 18.1 through 18.13, Tables 19.1 through 19.10, and Table 20. The fourth round testing results are summarized in Tables 21.1 through 21.9, Tables 22.1 through 22.8, Tables 23.1 through 23.8, and Tables 24.1 through 24.7. The concentrations of organic and inorganic compounds measured in the groundwater samples collected during the investigation that were found to exceed the regulatory standards are outlined in Table 25.

The groundwater samples collected during each quarterly sampling event were submitted to the selected state contract laboratory for the analysis of TCL VOCs (including MTBE, methane, ethane, and ethene), total TAL metals, and water chemistry parameters. The groundwater samples collected from monitoring well MW24 during the March/April 2003 and July 2003 sampling events were submitted for analysis of SVOCs. Due to a lack of SVOCs detected in the groundwater samples collected from monitoring well MW24, this parameter was deleted from the analytical testing list for the October 2003 and February 2004 sampling events. The groundwater samples collected during the July 2003 and February 2004 sampling events also included the analysis of "dissolved" metals. Discrete bottom samples of the groundwater/liquids contained in selected monitoring wells were collected during the July 2003, October 2003, and February 2004 sampling events to assess the presence/absence and chemical characteristics of separate-phase DNAPLs. Discrete top samples of the groundwater/liquids contained in selected monitoring wells were collected during the July 2003 sampling event to assess the presence/absence and chemical characteristics of separate-phase LNAPLs.

3.5.1   Concentrations of Volatile Organic Compounds

The groundwater samples collected from the monitoring well network during the investigation were found to contain the following VOCs: benzene, bromomethane, chloroethane, chloromethane, dissolved gases (methane, ethane, ethene), 1,1-dichloroethane (1,1-DCA), 1,1-dichloroethene (1,1-DCE), cis-1,2-dichloroethene (cis-1,2-DCE), 1,2-DCE (total), 1,4-dioxane, methyl tertiary butyl ether (MTBE), methylene chloride, 1,1,1-trichloroethane (1,1,1-TCA), trichloroethylene (TCE), and vinyl chloride (see

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Tables 9.1 through 9.10, Tables 12.1 through 12.9, Tables 18.1 through 18.13, and Tables 21.1 through 21.9).

Relatively low concentrations of acetone were detected in the groundwater samples collected from selected monitoring wells during each of the four quarterly sampling events. The concentrations of acetone detected in the groundwater samples were all below the PADEP Non-Residential Statewide Health-based Groundwater Standard. Acetone is a common solvent used by testing laboratories for sample preparation and the cleaning of equipment. Acetone is not considered to be a substance of environmental concern at the Bishop Tube site. Accordingly, the low concentrations of acetone detected in the groundwater samples collected from the monitoring wells during the investigation may be related to the sample preparation procedures performed by the testing laboratories.

Benzene and toluene were detected in trace amounts (i.e., "J" values) in the groundwater samples collected from selected monitoring wells during each of the four quarterly sampling events. The concentrations of benzene and toluene detected in the groundwater samples were all below the PADEP Non-Residential Statewide Health-based Groundwater Standards established for these compounds. Benzene and toluene are solvents commonly found in gasoline. These two solvents are not considered to be substances of environmental concern at the Bishop Tube site. The source of the benzene and toluene solvents measured in the groundwater samples collected from the monitoring well network is currently unknown, but may be related to a release of hydrocarbon compounds at the Mobil bulk oil terminal situated immediately west of the site.

During the March/April 2003 sampling event, bromomethane was detected at concentrations exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standard in monitoring wells MW02, MW12, and MW20. Concentrations of bromomethane in these groundwater samples were 67 μg/l, 280 μg/l, and 71 μg/l, respectively. Bromomethane is not considered to be a substance of environmental concern at the Bishop Tube site. Bromomethane is formed naturally in the ocean, probably by algae or kelp. This solvent is also manufactured for use as a soil or space fumigant to control vectors (i.e., mice/rats), insects, and fungi. The source of the bromomethane detected in the groundwater samples collected during the March/April 2003 sampling event is unknown.

Chloroethane was detected in monitoring well MW05 at a concentration of 4 μg/l during the July 2003 sampling event. This concentration of chloroethane was below the PADEP Non-Residential Statewide Health-based Groundwater Standard of 900 μg/l. Chloroethane is not considered to be a substance of environmental concern at the Bishop Tube site and was only detected in monitoring well MW05 during the July 2003 sampling event. The source of the chloroethane detected in the groundwater samples collected during the July 2003 sampling event is unknown. Chloroethane was used in the past as an additive in leaded gasoline. This information suggests that the chloroethane detected in the groundwater samples collected from monitoring well MW05 during the July 2003 sampling event may be related to a release of hydrocarbon compounds at the Mobil bulk oil terminal situated immediately west of the site.

A concentration of chloromethane (methyl chloride) of 50 μg/l that exceeded the PADEP Non-Residential Statewide Health-based Groundwater Standard of 3 μg/l, was detected in monitoring well MW21 during the October 2003 sampling event. Chloromethane is not considered to be a substance of environmental concern at the Bishop Tube site and was only detected one time during the investigation in monitoring well MW21. Based on information obtained from ATSDR (1999), chloromethane (methyl chloride) is released when wood, charcoal, and plastics are burned. Monitoring well MW21 is located along the east

side of the former incinerator room of Building #5 at the Bishop Tube site (see Figure 2). Remnants of burned materials may have been staged at this location until later removal. This supposition may provide an explanation for the elevated concentrations of chloromethane detected in the groundwater samples collected from monitoring well MW21 during the October 2003 sampling event.

Industrial solvents used in the degreasing and metal finishing industries are commonly formulated with additives to improve their performance. These additives, commonly referred to as stabilizers, serve to prevent solvent breakdown and inhibit reactions that may degrade the solvent properties. One ether-based stabilizer, 1,4-dioxane, has been added to chlorinated solvents (namely TCE and 1,1,1-TCA) in mixtures of 2% to 8% by volume. 1,4-dioxane is considered a VOC and has proven to be a compound of environmental concern at many chlorinated solvent release sites. Due to its high solubility, 1,4-dioxane is very mobile in the subsurface environment (similar to MTBE), and tends to migrate at the front of groundwater plumes containing chlorinated solvents. According to Mohr (2001), 1,4-dioxane is generally recalcitrant to biodegradation. Although considered a VOC, 1,4-dioxane is generally measured in groundwater samples using "SVOC" analytical methods rather than standard VOC techniques (i.e., lower detection limits can be obtained using the SVOC techniques rather than the VOC scan).

During the July 2003 sampling event, groundwater samples were collected from selected monitoring wells for the analysis of 1,4-dioxane. The analytical results for the July 2003 sampling event show that 1,4-dioxane was identified in the following three monitoring wells: MW25A (13 μg/l), MW28C (6 μg/l), and 30 Conestoga Road (8 μg/l). The concentrations of 1,4-dioxane measured in the groundwater samples were all below the PADEP Non-Residential Statewide Groundwater Standard of 24 μg/l. Monitoring wells MW25A and MW28C are situated north of the former degreaser area in Building #8 at the Bishop Tube site. The former vapor degreaser area in Building #8 is considered to be the principal source area for the chlorinated solvents contained in the underlying bedrock aquifer. The presence of 1,4-dioxane in monitoring wells MW25A and MW28C suggests that solvents containing this stabilizer were used in the past at the Bishop Tube facility. The well at 30 Conestoga Road is located approximately 1,600 feet northeast of the Bishop Tube site. The groundwater samples collected from the well at 30 Conestoga Road have been found to contain elevated concentrations of chlorinated solvents, including 1,1,1-TCA and TCE. The analytical results collectively suggest that: 1) solvents containing only small amounts of 1,4-dioxane were formerly used at the Bishop Tube site; and/or 2) the leading edge of the groundwater plume containing chlorinated solvents originating from the Bishop Tube site has migrated past the location of the well at 30 Conestoga Road. Based upon the high solubility and mobility of 1,4-dioxane in groundwater and the relatively high concentrations of chlorinated solvents detected in the groundwater samples collected from the well at 30 Conestoga Road, this latter supposition may better explain the distribution and migration of 1,4-dioxane and other chlorinated solvents comprising the groundwater plume. A further discussion regarding the migration of chlorinated solvents from the Bishop Tube site is presented in Section 5.5.2 of the report.

The analytical results show that relatively low concentrations (i.e., "J" values) of methylene chloride (dichloromethane) were measured/reported by the testing laboratories for the groundwater samples collected from selected monitoring wells throughout the investigation. Elevated concentrations of methylene chloride exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standard of 5 μg/l were detected in monitoring wells MW08 (13 μg/l), MW27A (18 μg/l), and MW27B (370 μg/l) during the February 2004, March/April 2003, and October 2003 sampling events, respectively. Monitoring wells MW08, MW27A, and MW27B are located either within or downgradient of the former drum storage area where drums containing new chemicals and wastes were stored for later use and

disposal, respectively. Methylene chloride is not considered to be a substance of environmental concern at the Bishop Tube site. Based on information obtained from HSIA (2003), methylene chloride is an active ingredient in paint removers. This compound is also used in metal coating and cleaning processes. It should be noted that methylene chloride is commonly used by environmental testing laboratories for sample preparation and the cleaning of testing equipment. This later information suggests that the source of the low concentrations of methylene chloride measured/reported in the groundwater samples collected during the investigation may be related to the sample preparation procedures performed by the testing laboratories. It is believed, however, that the source of the elevated concentrations of methylene chloride detected in the groundwater samples collected from monitoring wells MW08, MW27A, and MW27B during the February 2004, March/April 2003, and October 2003 sampling events is related to spills and leaks of chemicals/wastes in the former drum storage area.

MTBE was detected in the groundwater samples collected from the following monitoring wells during the investigation that exceeded the PADEP Non-Residential Statewide Health-based Groundwater Standard of 20 μg/l: MW03, MW05, MW06, MW08, MW09, MW11, MW12, MW20, MW24, MW25A, MW25B, MW25C, MW26A, MW27B, MW27C, MW28A, MW28B, and MW28C. The highest concentrations of MTBE were measured in the groundwater samples collected from monitoring wells MW25A (440 μg/l), MW25C (450 μg/l), and MW28C (960 μg/l) during the February 2004 sampling event. It should be noted that relatively high concentrations of MTBE were consistently detected in the groundwater samples collected from monitoring wells MW08, MW09, and MW20 during the investigation. MTBE is a petroleum additive that was not used in the industrial processes performed at the Bishop Tube site. The source of the elevated levels of MTBE in the groundwater samples collected from the monitoring well network is currently unknown, but may be related to a release of hydrocarbon compounds at the Mobil bulk oil terminal situated immediately west of the site.

Methane was detected in the following monitoring wells during the investigation: MW02, MW03, MW04, MW05, MW06, MW08, MW09, MW12, MW13, MW15, MW20, MW25A, MW25B, MW25C, MW26A, MW26B, MW26C, MW27B, MW27C, MW28A, MW28B, MW28C, and 30 Conestoga Road. The highest concentrations of methane were consistently detected in the groundwater samples collected from monitoring wells MW04, MW05, MW06, MW08, MW25A, MW25B, MW25C, MW28A, MW28B, and MW28C. Methane concentrations during the investigation were found to range from "not detected" to a high of 6,700 μg/l in monitoring well MW06 (October 2003 sampling event).

Methane is sometimes an indicator of anaerobic conditions and the presence of methanogenic bacteria. Methane can be produced from the microbiological breakdown of chloromethane or carbon dioxide in the subsurface environment (IRTC, 1998). In general, the monitoring wells that were found to contain the highest concentrations of methane (i.e., MW04, MW06, MW25A, MW25B, MW25C, MW28A, MW28B, and MW28C) also exhibited negative redox potentials.

Redox is short for reduction oxidation and is a measure of the oxidation capacity of the water. The redox potential (Eh) is a value defining the geochemical state of the water's capacity to donate (i.e., lose) or accept (i.e., gain) electrons. Groundwater typically contains both reduced and oxidized species. The following redox reactions will consume organic matter in groundwater: 1) aerobic degradation; 2) denitrification; 3) manganese (IV) reduction; 4) ferric iron reduction; 5) sulfate reduction; and 6) methane fermentation. The order of decreasing Eh conditions for these six reactions are as follows:

Reaction #1	Aerobic Degradation	+250 mV and higher
Reaction #2	Denitrification	+250 mV to +100 mV
Reactions #3 and #4	Iron and Manganese (IV) Reduction	+100 mV to 0 mV
Reaction #5	Sulfate Reduction	0 mV to -200 mV
Reaction #6	Methane Fermentation	-200 mV and lower

Water in contact with air will typically exhibit Eh values ranging from 350 mV to 500 mV. Microbically mediated redox processes may decrease the redox potential to values as low as –300 mV.

Ethane and ethene are generally considered metabolic end products of reductive dehalogenation of chlorinated solvents (ITRC, 1998). Accordingly, the presence of ethane and ethene may be used as indicators for the microbiologic degradation of chlorinated solvents. The analytical results show that ethane and ethene were consistently detected in the groundwater samples collected from monitoring wells MW05, MW06, MW08, and MW25B during the investigation. The highest concentrations of ethane and ethene were measured in the groundwater samples collected from monitoring wells MW05, MW06, and MW08.

As mentioned above, negative Eh values (i.e., low redox potentials) were recorded during the collection of groundwater samples from monitoring wells MW04, MW06, MW25A, MW25B, MW25C, MW28A, MW28B, and MW28C. It should be noted that the groundwater samples collected from these monitoring wells during the investigation were found to contain elevated concentrations of chlorinated solvents. The presence of methane, ethane, and ethene, as well as the negative redox potentials, suggests that naturally occurring microbes contained in the bedrock aquifer may be degrading (to a limited degree) the chlorinated solvents contained in the groundwater in the vicinity of monitoring wells MW04, MW06, MW25A, MW25B, MW25C, MW28A, MW28B, and MW28C. Based upon the high concentrations of methane, ethane, and ethene, naturally occurring microbes contained in the bedrock aquifer are also suspected to be degrading the chlorinated solvents contained in the groundwater in the vicinity of monitoring wells MW05, MW08, MW09, and MW20. The specific chemical processes responsible for the degradation and the rate of decay cannot be determined at this time based upon the limited amount of available data.

The following chlorinated solvents were detected in the groundwater samples collected from the monitoring well network during the investigation: 1,1-DCA, 1,1-DCE, cis-1,2-DCE, 1,2-DCE (total), PCE, 1,1,1-TCA, TCE, and vinyl chloride.

TCE was detected above the PADEP Non-Residential Statewide Health-based Groundwater Standard of 5 μg/l in all of the monitoring wells at the site with the exception of monitoring wells MW01 (upgradient well along the southern edge of the site) and MW24 (upgradient well along the western edge of the site). The highest concentrations of TCE (i.e., greater than 500 μg/l) were consistently measured in the groundwater samples collected from monitoring wells MW02, MW03, MW12, MW13, MW15, MW20, MW22, MW25A, MW25B, MW25C, MW26A, MW26B, MW26C, MW27B, MW28A, MW28B, MW28C, and the well at 30 Conestoga Road. The concentrations of TCE in these monitoring wells ranged from a low of 580 μg/l (MW20 in July 2003) to a high of 980,000 μg/l (MW26C in February 2004). Based on these concentrations, TCE is considered to be the chief substance of environmental concern in the groundwater at the Bishop Tube site. An evaluation of the concentration trends of TCE in comparison to groundwater levels is presented in Section 3.6 of the report. It should be noted that the TCE concentrations detected in the groundwater samples collected from monitoring wells MW02,

MW25A, MW25B, MW25C, MW26B, MW26C, MW27B, MW28A, MW28B, MW28C, and the well at 30 Conestoga Road were found to be greater than 1% of the pure-phase aqueous solubility limit for this solvent. As a general rule of thumb, Johnson and Pankow (1992) suggest that dissolved concentrations of TCE in groundwater equal to or greater than 1% of this solvent's pure-phase solubility limit (i.e., 11,000 µg/l) is an indication that separate-phase DNAPLs may be present in the subsurface. Separate-phase DNAPLs were observed in monitoring wells MW25A, MW25B, MW26A, MW26B, MW27B, MW28A, MW28B, and MW28C during the February 2004 sampling event. A further discussion regarding the occurrence of separate-phase DNAPLs in the fractured bedrock aquifer underlying the Bishop Tube site is presented in Section 3.7 of the report.

The analytical results show that PCE was detected in the groundwater samples collected from monitoring wells MW04, MW22, MW25A, MW25B, MW25C, MW26A, MW26C, MW27B, and MW28A during the investigation at concentrations exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standard of 5 µg/l (see Table 25). The highest concentrations of PCE were detected in the following monitoring wells during the investigation: MW25A (260 µg/l – October 2003), MW26C (1,600 µg/l – October 2003), and MW27B (190 µg/l – February 2004).

Elevated concentrations of 1,1,1-TCA exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standard of 200 µg/l were identified in the following monitoring wells during the investigation: MW02, MW03, MW11, MW12, MW13, MW15, MW16, MW21, MW25A, MW25B, MW25C, MW26A, MW26C, MW27B, MW28A, MW28B, MW28C, and the well at 30 Conestoga Road (see Table 25). The highest concentrations of 1,1,1-TCA were detected in the groundwater samples collected from monitoring wells MW02 (12,000 µg/l - October 2003), MW25B (4,800 µg/l – March/April 2003), MW25C (17,000 µg/l – July 2003), MW26A (6,900 µg/l - July 2003), MW28A (6,200 µg/l – March/April 2003), MW28B (6,800 µg/l - July 2003), and MW28C (6,200 µg/l – July 2003). Based upon the sampling results, 1,1,1-TCA was found to be the second most prevalent volatile organic compound contained in the groundwater at the Bishop Tube site. The quarterly groundwater sampling results indicate that elevated concentrations of 1,1,1-TCA have migrated offsite from the Bishop Tube facility to the northeast (toward the well at 30 Conestoga Road). The lateral extent of the 1,1,1-TCA concentrations contained in the groundwater plume has not been determined.

The analytical results for the groundwater samples collected during the quarterly sampling program indicate that elevated concentrations of 1,1-DCA, 1,1-DCE, cis-1,2-DCE, 1,2-DCE (total), and vinyl chloride exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standards were detected in the monitoring wells at the site (see Table 25). As will be discussed further in Section 4.0 of the report, 1,1-DCA, 1,1-DCE, cis-1,2-DCE, 1,2-DCE (total), and vinyl chloride are considered daughter products resulting from the breakdown of PCE, 1,1,1-TCA, and TCE.

The quarterly groundwater sampling results show that elevated concentrations of 1,1-DCA exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standard of 110 µg/l were detected in the groundwater samples collected from monitoring wells MW02, MW25A, MW25B, MW25C, MW28A, MW28B, and MW28C. During the investigation, the highest concentrations of 1,1-DCA were detected in the groundwater samples collected from the following monitoring wells: MW25A (1,300 µg/l – October 2003), MW25B (4,600 µg/l – October 2003), MW25C (4,000 µg/l – October 2003), and MW28C (4,000 µg/l – July 2003). It should be noted that these monitoring wells are all located immediately downgradient of the former vapor degreaser area in Building #8.

Elevated concentrations of 1,1-DCE exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standard of 7 μg/l were detected in the following monitoring wells during the investigation: MW02, MW05, MW09, MW11, MW12, MW13, MW15, MW16, MW25A, MW25B, MW25C, MW27B, MW28A, MW28B, MW28C, and the well at 30 Conestoga Road (see Table 25). During the investigation, the highest concentrations of 1,1-DCE were detected in the groundwater samples collected from monitoring wells MW25B (1,100 μg/l – July 2003), MW25C (1,200 μg/l – July 2003), MW28C (1,200 μg/l – October 2003), and 30 Conestoga Road (590 μg/l – July 2003). The analytical results for the well at 30 Conestoga Road indicate that the chlorinated solvent plume originating from the Bishop Tube site has migrated within the fractured bedrock aquifer at least 1,600 feet northeast of the site.

The quarterly groundwater sampling results show that elevated concentrations of 1,2-DCE (total) exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standard of 70 μg/l were detected in the groundwater samples collected from monitoring wells MW02, MW03, MW04, MW05, MW07, MW08, MW09, MW11, MW12, MW13, MW15, MW16, MW20, MW25A, MW25B, MW25C, MW26A, MW28A, MW28B, and MW28C (see Table 25). During the investigation, the highest concentrations of 1,1-DCE (total) were detected in the groundwater samples collected from the following monitoring wells: MW02 (1,200 μg/l – October 2003), MW12 (1,900 μg/l – October 2003), MW25A (1,800 μg/l – October 2003), MW25B (2,900 μg/l – February 2004), and MW25C (830 μg/l – July 2003). These monitoring wells are also located immediately downgradient of the former vapor degreaser area in Building #8.

Elevated concentrations of cis-1,2-DCE exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standard of 70 μg/l were detected in the following monitoring wells during the investigation: MW02, MW04, MW05, MW08, MW09, MW11, MW12, MW13, MW15, MW18, MW20, MW25A, MW25B, MW25C, MW28A, and MW28C (see Table 25). The highest concentrations of cis-1,2-DCE were detected in the groundwater samples collected from monitoring wells MW02 (1,100 μg/l – October 2003), MW12 (1,900 μg/l – October 2003), MW25A (1,800 μg/l – October 2003), MW25B (3,100 – February 2004), and MW25C (780 μg/l – February 2004). These monitoring wells are also located immediately downgradient of the former vapor degreaser area in Building #8. According to Chapelle and others (2003), cis-1,2-DCE is typically a by-product produced from the reductive dechlorination of TCE. These researchers suggest, as a general rule-of-thumb, that if the measured concentrations of cis-1,2-DCE comprise more than 80% of the total DCE levels (i.e., cis-1,2-DCE concentrations + trans-1,2-DCE) in a sample, the DCE is most likely of biogenic origin. It should be noted that only low to trace concentrations of trans-1,2-DCE were measured in the groundwater samples collected from the monitoring wells during the quarterly sampling program. Using the above referenced rule-of-thumb as a guideline, this information collectively suggests that the cis-1,2-DCE contained in the groundwater underlying the site may be related to the breakdown of PCE or TCE. Importantly, this finding indicates that some natural microbial dechlorination is occurring within the fractured bedrock aquifer underlying the study area.

Elevated concentrations of vinyl chloride exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standard of 2 μg/l were detected in the following monitoring wells during the investigation: MW04, MW05, MW06, MW07, MW08, and MW09 (see Table 25). During the investigation, the highest concentrations of vinyl chloride were detected in the groundwater samples collected from monitoring wells MW04 (40 μg/l – October 2003), MW05 (46 μg/l – February 2004), and MW08 (730 μg/l – October 2003). These monitoring wells are situated along the eastern (i.e., downgradient edge) of the Bishop Tube site. This information suggests that elevated concentrations of vinyl chloride may be migrating offsite

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within the underlying fractured bedrock aquifer. It should be noted that vinyl chloride was not detected in the groundwater samples collected from monitoring wells MW25A, MW25B, MW25C, MW26A, MW26B, MW26C, MW27B, MW28A, MW28B, MW28C, and the well at 30 Conestoga Road during the investigation. These monitoring wells were found to contain some of the highest concentrations of chlorinated solvents at the site. The absence of measurable levels of vinyl chloride in the groundwater samples collected from monitoring wells MW25A, MW25B, MW25C, MW26A, MW26B, MW26C, MW27B, MW28A, MW28B, MW28C, and 30 Conestoga Road is believed to be related to the high concentrations of other chlorinated solvent species in these wells. Based upon the high concentrations of the other chlorinated solvent species in these wells, the testing laboratories needed to perform a series of multiple dilutions on the groundwater samples. As a result, these multiple dilutions increased the minimum detection limit of individual compounds to a point that the testing laboratories could no longer quantify the concentration of vinyl chloride. Based upon the analytical results for the groundwater samples collected from monitoring wells MW04, MW05, MW06, MW07, MW08, and MW09, it is suspected that elevated concentrations of vinyl chloride exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standard of 2 μg/l exist in the fractured bedrock aquifer northeast of the Bishop Tube site.

Previous studies conducted at the Bishop Tube site (Baker, 2002a and 2003) have identified three principal source areas for the chlorinated solvents contained in the soils and groundwater. These three source areas are: the former vapor degreaser area in Building #5, the former vapor degreaser area in Building #8, and the former drum storage area. The location of these three principal source areas is shown on Figure 12. The highest concentrations (i.e., +1,000 μg/l) of chlorinated solvents have been detected in the groundwater samples collected from the following monitoring wells: MW02, MW03, MW12, MW13, MW15, MW20, MW22, MW25A, MW25B, MW25C, MW26A, MW26B, MW26C, MW27B, MW28A, MW28B, MW28C, and the well at 30 Conestoga Road. These monitoring wells are situated hydraulically downgradient to one of the three principal source areas referenced above. The groundwater samples collected from monitoring wells MW25A, MW25B, MW25C, MW26B, MW26C, MW27B, MW28A, MW28B, and MW28C have been found to contain the highest concentrations of TCE. The analytical results show that the concentrations of TCE increase with respect to depth in the underlying bedrock aquifer. According to Cohen and Mercer (1993), elevated concentrations of chlorinated solvents that increase with respect to depth suggest that separate-phase DNAPLs may be present in the subsurface. The concentration trends exhibited by TCE in the monitoring wells during the investigation suggest that separate-phase DNAPLs may locally exist in the deeper portion of the fractured bedrock aquifer underlying the Bishop Tube site. A further discussion regarding the potential occurrence of separate-phase DNAPLs in the fractured bedrock aquifer underlying the Bishop Tube site is presented in Section 3.7 of the report.

3.5.2   Concentrations of Volatile Organic Compounds Measured in Discrete Bottom Samples

Discrete bottom samples of the groundwater/liquids contained in selected monitoring wells were collected during the July 2003, October 2003, and February 2004 sampling events. These discrete bottom samples were collected to evaluate the presence/absence and chemical characteristics of separate-phase DNAPLs in the monitoring wells at the site. The discrete bottom samples for the July 2003 sampling event were collected prior to purging the monitoring wells (i.e., collected under static equilibrium conditions).

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Both "pre-purge" and "post-purge" discrete bottom samples were collected during the October 2003 and February 2004 sampling events for evaluating the presence/absence and chemical characteristics of separate-phase DNAPLs in the monitoring wells at the site. The "pre-purge" discrete bottom samples were collected to evaluate the equilibrium conditions in the monitoring wells. The "post-purge" samples were collected to determine if any separate-phase liquids may have been drawn into monitoring wells by the well purging (i.e., pumping) activities performed to collect the standard quarterly groundwater samples.

During the July 2003 sampling event, discrete bottom samples were collected from the following monitoring wells: MW02, MW03, MW12, MW13, MW15, MW18, MW20, MW22, MW25A, MW25B, MW25C, MW26A, MW26B, MW26C, MW27A, MW27B, MW28A, MW28B, MW28C, and the well located at 30 Conestoga Road. The analytical results for the July 2003 sampling event show that elevated concentrations of 1,1-DCA, 1,1-DCE, cis-1,2-DCE, 1,2-DCE (total), PCE, 1,1,1-TCA, and TCE, exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standards established for these parameters, were identified in the discrete bottom samples collected from the selected monitoring wells at the site (see Tables 16.1 through 16.5). The highest concentrations of TCE were identified in the discrete bottom samples collected from monitoring wells MW22, MW25A, MW25B, MW25C, MW26B, and MW26C at 78,000 μg/l, 42,000 μg/l, 46,000 μg/l, 120,000 μg/l, 530,000 μg/l, and 410,000 μg/l, respectively. It should be noted that although elevated concentrations of TCE were detected in the discrete bottom groundwater samples, no separate-phase DNAPLs were identified in the monitoring wells during the July 2003 sampling event.

During the October 2003 sampling event, "pre-purge" and "post-purge" discrete bottom samples were collected from monitoring well MW26C. The analytical results for the October 2003 sampling event show that the concentrations of TCE detected in the "pre-purge" bottom sample, low-flow groundwater sample (collected under pumping conditions), and "post-purge" bottom sample collected from monitoring well MW26C were 580,000 μg/l, 800,000 μg/l, and 980,000 μg/l, respectively. It should be noted that the concentration of TCE detected in the "post purge" bottom sample collected from monitoring well MW26C was 38% higher than the low-flow groundwater sample (pumping conditions). According to Cohen and Mercer (1993), a decrease in dissolved concentrations during pumping conditions with a significant increase after the pumps are turned off suggests that separate-phase DNAPLs may be present in the vicinity of a well. Based upon this criterion, the large increase in TCE concentrations between the low-flow (i.e., pumping) groundwater sample and the "post-purge" bottom sample suggests that separate-phase DNAPLs (source for dissolved TCE concentration) may be present in the fractures of the bedrock in the immediate vicinity of monitoring well MW26C. It should be noted that although elevated concentrations of TCE were detected in the low-flow (i.e., pumping) groundwater sample and the "post-purge" bottom sample, no separate-phase DNAPLs were identified in the monitoring well MW26C during the October 2003 sampling event.

During the February 2004 sampling event, "pre-purge" and "post-purge" discrete bottom samples were collected from the following monitoring wells at the site: MW02, MW12, MW22, MW25C, MW26A, MW26B, MW26C, MW27B, MW28A, MW28B, MW28C, and the well located at 30 Conestoga Road. The analytical results for the February 2004 sampling event show that elevated concentrations of 1,1-DCA, 1,1-DCE, cis-1,2-DCE, 1,2-DCE (total), MTBE, PCE, 1,1,1-TCA, and TCE, exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standards established for these parameters, were identified in the "pre-purge" discrete bottom samples collected from the selected monitoring wells at the site (see Tables 24.1 through 24.6). The highest concentrations of TCE were identified in the "pre-purge"

discrete bottom samples collected from monitoring wells MW22, MW25C, MW26B, and MW26C at 100,000 μg/l, 36,000 μg/l, 170,000 μg/l, and 580,000 μg/l, respectively. It should be noted that the concentrations of TCE detected in the "post-purge" bottom samples were higher in comparison to the low-flow groundwater sample (pumping conditions) in the following monitoring wells: MW02 (17% increase), MW22 (90% increase), MW25C (18% increase), and MW28C (49% increase). As discussed above, a decrease in dissolved concentrations during pumping conditions with a significant increase after the pumps are turned off suggests that separate-phase DNAPLs may be present in the vicinity of a well. Based upon this criterion, the large increase in TCE concentrations between the low-flow (i.e., pumping) groundwater sample and the "post-purge" bottom sample suggests that separate-phase DNAPLs (source for dissolved TCE concentration) may be present in the fractures of the bedrock in the immediate vicinity of monitoring wells MW02, MW22, MW25C, and MW28C. It should be noted that separate-phase liquids were observed in monitoring well MW28C during the February 2004 sampling event. The reader is referred to Section 3.7 of the report, for a detailed discussion regarding the assessment of DNAPL occurrence in the fractured bedrock aquifer underlying the study area.

3.5.3   Concentrations of Volatile Organic Compounds Measured in Discrete Top Samples

Discrete top samples of the groundwater/liquids contained in selected monitoring wells were collected during the July 2003 sampling event. These discrete top samples were collected to evaluate the presence/absence and chemical characteristics of separate-phase LNAPLs floating on top of the groundwater contained in monitoring wells MW04, MW05, MW06, MW08, MW09, and MW24. The discrete top samples for the July 2003 sampling event were collected prior to purging the monitoring wells (i.e., collected under static equilibrium conditions).

The analytical results show that the top groundwater samples collected from monitoring wells MW04, MW05, MW08, and MW09 contained elevated concentrations of 1,2-DCE (total), TCE, and vinyl chloride that exceeded the PADEP Non-Residential Statewide Health-based Groundwater Standards. The highest concentration of TCE was identified in monitoring well MW09 at 760 μg/l. The highest concentration of vinyl chloride was identified in monitoring well MW08 at 230 μg/l. Elevated concentrations of 1,1-DCA exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standard were detected in monitoring wells MW04, MW05, and MW06. 1,1,1-TCA was detected at levels above the PADEP Non-Residential Statewide Health-based Groundwater Standard in monitoring wells MW08 and MW09 (see Tables 17.1 and 17.2). It should be noted that no separate-phase LNAPLs were observed floating on the top of the groundwater in the monitoring wells during the July 2003 sampling event. An oily sheen and petroleum odor, however, was noted in monitoring well MW24. This monitoring well is situated along the western edge of the Bishop Tube property in close proximity to the Mobile bulk oil facility.

3.5.4   Concentrations of Semi-Volatile Organic Compounds

The groundwater samples collected from monitoring well MW24 during the March/April 2003 and July 2003 sampling events were analyzed for SVOCs. The analytical results show that no SVOCs were detected in the groundwater samples collected from monitoring well MW24 that exceeded the PADEP Non-Residential Health-based Groundwater Standards during the March/April 2003 sampling event (see Table 10).

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The groundwater samples collected from monitoring well MW24 during the July 2003 sampling event, however, show that the measured concentration of bis(2-ethyl-hexyl)phthalate (10 μg/l) exceeded the PADEP Non-Residential Statewide Groundwater Health-based Standard of 6 μg/l (see Tables 13.1 and 13.2). It should be noted that monitoring well MW24 was constructed using PVC riser and well-screen materials. Bis(2-ethyl-hexyl)phthalate is typically used as a "plasticizer" to increase the flexibility of plastic materials (Francis, 1993). This information suggests that the concentrations of bis(2-ethyl-hexyl)phthalate detected in the groundwater samples collected from monitoring well MW24 may be related to the PVC riser and wellscreen materials rather than extraneous compounds contained in the groundwater at the Bishop Tube site. Due to the low concentration of SVOCs detected in the groundwater samples collected from monitoring well MW24 in March/April 2003 and July 2003, this parameter was deleted from the analytical testing list for the October 2003 and February 2004 sampling events.

3.5.5   Concentrations of Total and Dissolved Metals

The groundwater samples collected from the monitoring wells during each quarterly sampling event were submitted to the selected state contract laboratory for the analysis of total TAL metals. The groundwater samples collected during the July 2003 and February 2004 sampling events were also submitted to the selected state contract laboratory for the analysis of dissolved metals. It should be noted that only the results for "dissolved" heavy metal concentrations were used to evaluate water quality impacts to the aquifer underlying the project area.

The analytical results show that aluminum, calcium, iron, magnesium, manganese, and sodium were detected in the groundwater throughout the site. Some of the aluminum, iron, and magnesium concentrations were found to exceed the PADEP "Secondary" Drinking Water Standards established for these compounds. Sloto (1987) and Senior and others (1997) report that these metals are typical constituents of the host bedrock (i.e., Conestoga formation). Based on this information, the elevated total concentrations of aluminum, calcium, iron, magnesium, manganese, and sodium detected in the monitoring wells throughout the study area are believed to be related to the natural conditions occurring in the bedrock aquifer underlying the East Whiteland Township, Chester County area.

Antimony, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, lead, mercury, nickel, silver, thallium, vanadium, and zinc were all detected at relatively low levels in the "total" and "dissolved" groundwater samples across the site during the investigation. The "total" concentrations of antimony, arsenic, cadmium, cobalt, copper, mercury, silver, vanadium, and zinc detected in the groundwater samples were all below the PADEP Non-Residential Statewide Health-based Groundwater Standards established for these parameters. Beryllium was detected at concentrations of 6.3 μg/l (MW25B - March/April 2003 sampling event) and 9.7 μg/l (MW28C – February 2004 sampling event), exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standard of 4 μg/l. Barium was detected in the groundwater sample collected from monitoring well MW28C at a concentration of 3,290 during the μg/l, exceeding the exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standard of 2,000 μg/l (see Table 25). Barium and beryllium are not considered to substances of environmental concern at the Bishop Tube site.

The highest concentrations of "total" lead were detected in monitoring wells MW11, MW24, and MW28C during the investigation. These monitoring wells are situated along the western and northern edges of the Bishop Tube site. It should be noted that a release of hydrocarbon compounds has occurred at the Mobil bulk oil terminal situated immediately west of the Bishop Tube site. Petroleum products have

been stored at this bulk oil terminal since at least 1947 (based upon the review of historical aerial photographs, [Baker, 2002a, and 2002b]). Although the source of the elevated concentrations of "total" lead in monitoring wells MW11, MW24 and MW28C is unknown, the above referenced historical information suggests that the lead in these wells may be related to the former release of leaded gasoline at the neighboring Mobil bulk oil terminal. This supposition is supported by the elevated concentrations of MTBE (a gasoline additive) detected in the groundwater samples collected from monitoring wells MW11 and MW24 during the investigation.

Dissolved concentrations of chromium, nickel, and thallium exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standards were detected in the groundwater samples collected from several monitoring wells (see Table 25). Specifically, "dissolved" concentrations of chromium exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standard of 100 µg/l were detected in the following monitoring wells during the investigation: MW04, MW07, MW21, MW22, MW23, MW25C, and MW27A. The monitoring wells containing the highest concentrations of "dissolved" chromium include: MW22 (734 µg/l – July 2003 sampling event), MW23 (427 µg/l – February 2004 sampling event), MW27A (451 µg/l – July 2003 sampling event), and MW27B (528 µg/l – February 2004 sampling event).

Elevated concentrations of "dissolved" nickel that exceeded the PADEP Non-Residential Statewide Health-based Groundwater Standard of 100 µg/l were detected in the following monitoring wells during the investigation: MW04, MW07, MW21, MW22, MW23, MW27A, and MW27B (see Table 25). The monitoring wells containing the highest concentrations of "dissolved" nickel include: MW04 (186 µg/l – July 2003 sampling event), MW07 (153 µg/l – July 2003 sampling event), MW21 (138 µg/l – July 2003 sampling event), and MW22 (345 µg/l – February 2004 sampling event).

Dissolved concentrations of thallium were detected in the groundwater samples collected from the from following monitoring wells during the July 2004 sampling event that exceeded the PADEP Non-Residential Statewide Health-based Groundwater Standard of 2 µg/l: MW11 (5.5 µg/l), MW21 (5.1 µg/l), MW26A (5.6 µg/l), and the well at 30 Conestoga Road (5.8 µg/l).

Elevated concentrations of "dissolved" lead that exceeded the PADEP Non-Residential Statewide Health-based Groundwater Standard of 5 µg/l were detected in monitoring well MW09 during the February 2004 sampling event.

In general, the wells containing the highest "total" and "dissolved" concentrations of heavy metals are located along the eastern portion of the Bishop Tube site in the vicinity of the former drum storage area, the former cesspool area (i.e., east of the incinerator room in Building #5 near monitoring well MW21), and the former waste lagoon (i.e., southeastern corner of Building #8). Based upon historical records of the former plant operating procedures, heavy metal liquid wastes and sludges were processed, stored, and/or disposed of in each of these areas. This information suggests that the source of the heavy metals contained in the groundwater along the eastern portion of the Bishop Tube site may be related to the former industrial and waste-handling procedures performed at the facility.

3.5.6   Water Quality Parameters and Inorganic Constituents

The groundwater samples collected from the monitoring wells during each quarterly sampling event were submitted to the selected state contract laboratory for the analysis of the following water quality

parameters: bicarbonate alkalinity, carbonate alkalinity, chloride, fluoride, nitrate, nitrite, total kjeldahl nitrogen, sulfate, sulfite, total dissolved solids (TDS), and total organic carbon (TOC).

Elevated concentrations of fluoride that exceeded the PADEP Non-Residential Statewide Health-based Groundwater Standard of 2 mg/l were detected in the following monitoring wells during the investigation: MW04, MW05, MW06, MW07, MW08, MW09, MW18, MW21, MW22, MW23, MW25A, MW26A, MW26B, MW27A, and MW27B (see Table 25). The monitoring wells containing the highest concentrations of fluoride include: MW04 (16.6 mg/l – March/April 2003 sampling event), MW21 (21 mg/l – March/April 2003 sampling event), MW22 (20.6 mg/l – March/April 2003 sampling event), and MW27A (16.1 mg/l – July 2003 sampling event). It should be noted that fluoride concentrations generally decrease with respect to depth within the fractured bedrock aquifer underlying the site. This general trend is believed to be a result of the interaction of fluoride and the carbonate minerals comprising the underlying fractured bedrock aquifer.

The analytical results show that the concentration of nitrate measured in the groundwater sample collected from monitoring well MW07 (10.1 mg/l) during the July 2003 sampling event exceeded the PADEP Non-Residential Statewide Health-based Groundwater Standard of 10 mg/l.

The monitoring wells containing the highest concentrations of fluoride and nitrate are located along the eastern portion of the Bishop Tube site in the vicinity of the former drum storage area, the former cesspool area (i.e., east of the incinerator room in Building #5 near monitoring well MW21), and the former waste lagoon (i.e., southeastern corner of Building #8). Based upon historical records of the former plant operating procedures, hydrofluoric and nitric acid pickle liquor was processed, stored, and/or disposed of in each of these areas. This information suggests that the source of the fluoride and nitrate contained in the groundwater along the eastern portion of the Bishop Tube site is most likely related to the former industrial and waste handling procedures performed at the facility.

In May 2003, representatives from the PADEP conducted a detailed surface water study of Little Valley Creek (approximately 1.5 miles downstream from the Bishop Tube site). This study included the collection and analysis of 24 discrete surface water samples from Little Valley Creek, un-named tributary streams, and springs. The analytical results showed that the surface water samples collected from Little Valley Creek adjacent to the Bishop Tube site contained concentrations of fluoride that were anomalously higher than the background samples collected upstream of the facility. The concentrations of fluoride contained in the surface water of Little Valley Creek were found to decrease downstream of the Bishop Tube site. As noted in Section 3.3 of the report, shallow groundwater underlying the eastern edge of the Bishop Tube site is believed to be serving as a source of baseflow to Little Valley Creek. This finding suggests that the source of the anomalously high concentrations of fluoride contained in the surface water of Little Valley Creek is most likely related to the discharge of shallow groundwater via baseflow from the Bishop Tube site.

3.6   Trend Analysis of the Chlorinated Solvent Groundwater Plume

The analytical results show that elevated concentrations of TCE exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standards were detected in the groundwater samples collected from each monitoring well at the Bishop Tube site with the exception of MW01 and MW24. The groundwater samples collected during the investigation show that the highest concentrations of TCE were detected on a regular basis in monitoring wells MW22, MW25A, MW25C, MW26A, MW26C, MW28A, MW28C,

and the well located at 30 Conestoga Road (see Table 25). To make a determination regarding the migration of the groundwater plume in these wells, a comparison was made between: 1) changes in the concentration of TCE with respect to fluctuations of groundwater levels; and 2) changes in the concentration of TCE over time. This comparison was used to evaluate whether the concentrations in a well were increasing, decreasing, or stable.

To determine if a relationship exists between changes in the measured concentration of TCE and groundwater fluctuations, graphs were constructed using the measured values of these parameters for monitoring wells MW22, MW25A, MW25C, MW26A, MW26C, MW28A, MW28C, and the well located at 30 Conestoga Road (see Appendix E). The visual comparison of the graphs to determine whether a rise/decrease in the concentration of TCE corresponds to a rise/decrease in the elevation of the groundwater is a "qualitative" judgment. To "quantitatively" evaluate whether a relationship exists between the measured concentrations of TCE and the fluctuation of groundwater levels in a well, a statistical procedure was used to analyze the testing data. The basis for using this approach assumed that some type of linear relationship would exist between the changes in the concentration of TCE and the fluctuation of the groundwater levels. Johnson (1976) reports that the coefficient of linear correlation (R) is the measure of the strength of the linear relationship between two variables. This strength of relationship is determined by the amount of effect any change in one variable has on the other. To determine the strength of relationship, a graph is created by plotting the measurements of one variable against the other. Perfect linear correlation would arise where all the plotted points fall exactly on a straight line. This could be either a positive correlation or a negative correlation, depending on the slope of the line (increasing or decreasing).

The data points for two variables plotted on a graph often do not show a perfect linear relationship. To reduce the level of subjective judgment needed for assessing whether a linear relationship exists, a numeric value can be calculated using the raw data values to determine the strength of relationship between the two variables. This strength of relationship is referred to as the linear correlation coefficient (R). The linear correlation coefficient (R) exhibits values ranging between −1 and +1 (Johnson, 1976). A negative correlation (i.e., increase in one variable with an offsetting decrease in the other variable) would be indicated when the R-value is closer to −1. Conversely, a positive correlation (i.e., an increase/decrease in one variable with a corresponding increase/decrease in the other variable) would be indicated when the R-value is closer to +1. The strength of relationship between the two variables (either positive or negative) would increase the closer the linear correlation coefficient (R) values are to +1 and −1, respectively. A lack of relationship would be indicated when the linear correlation coefficient (R) values approach zero. To determine the strength of relationship between the TCE concentrations and the groundwater fluctuations recorded in monitoring wells MW22, MW25A, MW25C, MW26A, MW26C, MW28A, MW28C, and the well located at 30 Conestoga Road, the linear correlation coefficient (R) was calculated using the following formula referred to as Pearson's product moment:

$$R = \frac{n(\sum xy) - (\sum x)(\sum y)}{\sqrt{n(\sum x^2) - (\sum x)^2} \bullet \sqrt{n(\sum y^2) - (\sum y)^2}} \qquad \text{Equation 1}$$

where:

        R =   Pearson Product Moment correlation value

        n =   number of observations

        Σx =   sum of x values (groundwater elevations)

Σy =   sum of y values (TCE concentrations)
Σxy =   sum of the x values multiplied with the y values

A value for the coefficient of linear correlation (R) was calculated for monitoring wells MW22 (-0.78), MW25A (-0.74), MW25C (+0.79), MW26A (-0.92), MW26C (-0.66), MW28A (-0.35), MW28C (+0.66), and the well located at 30 Conestoga Road (-0.01). The calculations and graphs used to determine the coefficient of linear correlation (R) for these monitoring wells are presented in Appendix E.

Based on these values, the calculations show that a good positive correlation exists between the measured concentrations of TCE and the fluctuations of groundwater levels in monitoring wells MW25C and MW28C. This correlation suggests that a rise in groundwater levels produces an increase in the dissolved TCE concentrations. Conversely, a drop in groundwater levels produces a decrease in the dissolved TCE concentrations. The reason for this cause and effect in monitoring wells MW25C and MW28C is unknown. Cohen and Mercer (1993) point out that the dissolution process of residual DNAPLs can be influenced by varying hydraulic conditions (i.e., changes in groundwater flow directions or rates) in the subsurface. Accordingly, localized groundwater flow changes within the bedrock aquifer may provide an explanation for the good positive correlation between changes in the measured concentrations of TCE and groundwater levels in monitoring wells MW25C and MW28C.

It should be emphasized that the groundwater levels measured in monitoring well MW28C in March 2003 were anomalously lower in elevation than those recorded in July 2003, October 2003, and February 2004. The cause(s) for the anomalously lower groundwater elevations in monitoring well MW28C are unknown, but could be related: 1) remnant effects from the construction of monitoring well MW28C in March 2003; and/or 2) drawdown effects created by a nearby pumping well. Baker is not aware of any pumping wells in the vicinity of the Bishop Tube site. Based upon the stability of the water levels during the period between July 2003 and February 2004, the lower water levels recorded in monitoring well MW28C in March 2003 are attributed to remnant effects related to the well construction procedures.

Based on the calculated coefficient of linear correlation (R) values, a good negative correlation was found between the measured concentrations of TCE and the fluctuations of groundwater levels in monitoring wells MW22, MW25A, MW26A, MW26C, and MW28A. This correlation suggests that a rise in groundwater levels produces a decrease in the dissolved TCE concentrations. Conversely, a drop in groundwater levels produces an increase in the dissolved TCE concentrations. The reason for this cause and effect in monitoring wells MW22, MW25A, MW26A, MW26C, and MW28A may be due to the effects of dilution within the fractured aquifer. This supposition suggests that at times when groundwater levels are high (larger/faster rate of groundwater flow), the TCE concentrations within the fractured bedrock aquifer underlying the study area reduced/diluted.

Based upon the TCE concentrations and static water level data, no correlation between these parameters could be made for the well at 30 Conestoga Road.

To evaluate the plume stability in monitoring wells MW22, MW25A, MW25C, MW26A, MW26C, MW28A, MW28C, and the well located at 30 Conestoga Road, graphs of the natural log of the TCE concentrations versus time were created for the data collected from each of these wells. Based upon the position of the data points, a best-fit straight line was plotted on each graph. The slope of this line defines the concentration trend in each monitoring well. A slope less than zero indicates that the concentrations of the compound of interest are decreasing. Conversely, a positive slope indicates that the concentrations of

the compound of interest are increasing. A slope close to or equal to zero suggests that the concentrations of the compound of interest have stabilized in a well. To reduce the level of subjective judgment in drawing the best-fit line on each graph, a numeric value for the slope can be calculated using the raw data values. The following equation was used to calculate the best-fit straight line for the raw data collected for monitoring wells MW22, MW25A, MW25C, MW26A, MW26C, MW28A, MW28C, and the well located at 30 Conestoga Road:

$$\boldsymbol{Slope} \; = \; \frac{n\sum xy - \left(\sum x\right)\left(\sum y\right)}{n\sum x^2 - \left(\sum x\right)^2} \qquad \text{Equation 2}$$

where:

Slope = calculated slope of a best fit straight line based upon data values
n = number of observations
Σx = sum of the x values (number of days for observation period)
Σy = sum of the y values (natural log of TCE concentrations)
Σxy = sum of the x values multiplied with the y values

The following values for the best-fit straight lines were determined for monitoring wells MW22, MW25A, MW25C, MW26A, MW26C, MW28A, MW28C, and the well located at 30 Conestoga Road: +0.00043, -0.00003, -0.00029, -0.0015, +0.00108, +0.00087, +0.00196, and +0.00026, respectively (see Appendix E for graphs and calculations). A stable to decreasing trend was determined for the TCE concentrations measured in the groundwater samples collected from monitoring wells MW25A, MW25C, and MW26A. It should be noted that monitoring wells MW25A/MW25C and MW26A are situated downgradient of the former vapor degreaser area in Building #8 and the former drum storage area, respectively (i.e., two suspected source areas for the chlorinated solvents identified in the groundwater at the Bishop Tube site). Monitoring well MW26A is situated at the northeast corner of the Bishop Tube property. The stable to decreasing trend for the TCE concentrations measured in the groundwater samples collected from monitoring wells MW25A, MW25C, and MW26A suggests that the center of mass for the groundwater plume may have already migrated past the intervals of these wells.

Conversely, the concentrations of TCE were found to be increasing in monitoring wells MW22, MW26C, and MW28C. The strongest positive trends were detected in monitoring wells MW26C and MW28C, located at the northeast corner of the Bishop Tube property and north of Building #8, respectively. The screened intervals of monitoring wells MW26C and MW28C monitor the deep groundwater flow system underlying the site. The former vapor degreaser area located in Building #8 is suspected to be the one of the principal source areas for the TCE contained in the bedrock aquifer underlying the Bishop Tube site. The strong positive slope indicated by the trend analysis for monitoring wells MW26C and MW28C suggests that residual non-aqueous phase liquids are still leaching to the groundwater and possibly migrating within the fractured bedrock aquifer at these two well locations. The positive slopes indicated by the trend analyses for monitoring wells MW26C and MW28C suggests that TCE is continuing to migrate within the deeper portion of the fractured bedrock aquifer in the vicinity of these screened intervals. In addition, the trend analyses for monitoring wells MW28A and the well located at 30 Conestoga Road indicate that the TCE plume in the shallow groundwater system is continuing to migrate in a north-northeasterly direction. These findings are consistent with the results of the RT3D groundwater flow model showing that the chlorinated solvent groundwater plume is migrating in a north-northeast direction from the Bishop Tube site (see Section 5.5.2 of the report).

3.7 <u>Assessment of Potential DNAPL Occurrence in the Bedrock Aquifer</u>

Guidelines published by Cohen and Mercer (1993) suggest that separate-phase DNAPLs may be present in the subsurface if the groundwater concentrations are greater than 1% of the pure-phase aqueous solubility of the component of interest. Accurate use of the 1% solubility rule requires additional information regarding the parent DNAPL composition such that mole fractions can be determined. If the DNAPL is comprised of one component, the mole fraction of this component can be approximated to be one. If the parent DNAPL is comprised of several compounds, however, a modified approach is required to assess the presence of separate-phase DNAPLs based upon groundwater concentrations.

The compounds comprising a multi-component DNAPL will not dissolve into groundwater at their published solubility values, but rather will compete with one another in the dissolution process. The dissolution of a multi-component DNAPL in groundwater can be described by Raoult's Law. Raoult's Law states that the effective solubility of a NAPL component in groundwater is equal to the product of the mole fraction in the NAPL and the single aqueous solubility of the compound. Raoult's law can therefore be expressed as:

$$C_i = m_i \bullet S_i \qquad \text{Equation \#3}$$

where:

C_i = is the effective solubility of component i.
m_i = is the mole fraction of component i in the NAPL.
S_i = is the single component solubility of component i.

In practical terms, the effective solubility of a compound is the maximum concentration that could possibly be achieved in groundwater. In regards to the 1% solubility rule, it is 1% of the "effective" solubility that is taken as a possible indicator of DNAPL presence, rather than using just 1% of the single component solubility. Importantly, the principals of Raoult's Law suggest that separate-phase DNAPLs may be present in the subsurface environment even though the groundwater concentration of each NAPL component is less than 1% of a single component's pure-phase solubility. This is based upon the competition between each NAPL component in the dissolution process. Accordingly, consideration of the cumulative mole fractions is required to more fully evaluate the presence/absence of separate-phase DNAPLs using groundwater concentrations.

The concentrations of compounds measured in groundwater samples may not accurately represent the subsurface conditions based upon the effects of: 1) borehole dilution; 2) hydrodynamic dispersion; 3) non-optimal well placement (i.e., outside the center of the plume); and 4) abiotic/biotic degradation. According to Kueper and others (2003), the degree of dilution can be represented by the parameter a, such that:

$$a = \frac{C_i^{obs}}{C_i} \qquad \text{Equation \#4}$$

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where:

C_i^{obs} = is the concentration observed in the monitoring well.

C_i = is the effective solubility given by Equation #3.

For a multi-component DNAPL, the sum of the mole fractions must equal one, in accordance with the following relationship:

$$\sum \frac{C_i}{S_i} = 1 \qquad \text{Equation \#5}$$

Combining Equations #4 and #5 yields:

$$\sum \frac{C_i^{obs}}{S_i} = a \qquad \text{Equation \#6}$$

If there are no effects attributable to borehole dilution, dispersion, degradation, and monitoring well placement, parameter a would be equal to one in the vicinity of the multi-component DNAPL. Because these processes typically occur to some degree in the subsurface environment, the calculation of parameter a will generally be less than one. According to Kueper and others (2003), if the 1% effective solubility rule is used, DNAPL may be present in the subsurface if parameter $a > 0.01$ (i.e., 1%, representing the effective solubility of all compounds dissolved in the groundwater). This supposition is based upon the assumption that the effects of borehole dilution, dispersion, and degradation for each NAPL component dissolved in the groundwater are influenced equally in the subsurface environment. In regards to the effects of borehole dilution and dispersion, this may be a reasonable assumption. It should be noted that the effects of degradation may be different for each NAPL component dissolved in the groundwater (i.e., some compounds may preferentially degrade over others). For monitoring wells situated in close proximity to the DNAPL source, however, the amount of degradation may be minimal such that the calculation of parameter a by Equation #6 will provide reasonable results for assessing the presence/absence of separate-phase DNAPLs in the subsurface.

The analytical results for the groundwater samples collected during the July 2003, October 2003, and February 2004 sampling events were used to assess the presence/absence of separate-phase DNAPLs in the vicinity of the onsite and offsite monitoring wells. The March/April 2003 sampling results were not used for this assessment because this sampling event was conducted shortly after the drilling/well development procedures were performed in monitoring wells MW25, MW26, MW27, and MW28. In making the assessment of the potential occurrence of DNAPLs within the fractured bedrock aquifer based upon the concentrations of VOCs contained in the groundwater, it was deemed reasonable to use the analytical results representing static/equilibrium conditions rather than those collected immediately after the drilling/well development procedures. The calculations used to determine parameter a by Equation #6 for the July 2003, October 2003, and February 2004 sampling results are outlined in Appendix F. A summary of the values determined for parameter a and the assessment of the potential occurrence of separate-phase DNAPLs based upon the groundwater concentrations are summarized in Table 26. It should be noted that the calculations were only performed for selected background monitoring wells and

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target wells within or downgradient to the principal source areas at the site. The monitoring wells evaluated in this assessment included: MW01, MW02, MW03, MW12, MW15, MW22, MW23, MW25A, MW25B, MW25C, MW26A, MW26C, MW27B, MW28A, MW28B, MW28C, and the well at 30 Conestoga Road.

Based upon the relatively low concentrations of VOCs detected in the groundwater samples collected from monitoring wells MW01, MW03, MW12, MW15, MW22, MW23, and MW26A separate-phase DNAPLs are probably <u>not</u> <u>present</u> in the fractured bedrock aquifer in the vicinity of these wells (see Table 26). The calculations, however, show that separate-phase DNAPLs may be situated in the fractured bedrock aquifer in the vicinity of the following monitoring wells: MW02, MW25A, MW25B, MW25C, MW26B, MW26C, MW27B, MW28A, MW28B, MW28C, and the well at 30 Conestoga Road (see Table 26).

It should be noted that during the July 2003 and February 2004 sampling events, bottom samples were collected from selected monitoring wells using a discrete interval sampler for determining the presence/absence of separate-phase DNAPLs. A portion of the fluids collected from the bottom of the monitoring wells during the February 2004 sampling event was retained by Baker for further inspection and analysis. Cohen and Mercer (1993) report that many DNAPL compounds will fluoresce in the present of ultra-violet light. To verify the presence/absence of DNAPLs in the monitoring wells at the Bishop Tube site, the discrete bottom samples collected by Baker in February 2004 were checked for phase separation and fluorescence. The results of Baker's observations of the discrete bottom samples collected from the monitoring wells during the February 2004 sampling event are presented in Table 27. The inspection of the discrete bottom samples showed that the groundwater collected from monitoring wells MW25A, MW25B, MW26A, MW26B, MW27B, MW28A, MW28B, and MW28C contained separate-phase liquids exhibiting a dull-pale yellow fluorescence when inspected under ultra-violet light. It should be noted that the absence of separate-phase liquids in the groundwater samples collected from monitoring wells MW02, MW25C, and MW26C may be related to the sampling handling/processing techniques. Specifically, when the discrete interval sampler was extracted from each monitoring well, the sample vials to be submitted to the testing laboratory were filled first. The residual fluids contained in the discrete interval sampler were then retained by Baker for in-house inspection to determine the presence/absence of separate-phase liquids. Based upon this procedure, the residual fluids contained in the discrete interval sampler following the collection of the samples for laboratory analysis may or may not have contained separate-phase liquids. The results of the visual inspection of the bottom samples collected during the February 2004 sampling event support the findings outlined above regarding the cumulative mole fraction analysis, indicating that separate-phase DNAPLs are present in the bedrock aquifer underlying the Bishop Tube site in the vicinity of monitoring wells MW25A, MW25B, MW26A, MW26B, MW27B, MW28A, MW28B, and MW28C.

The source of the DNAPLs occurring in the fractured bedrock aquifer in the vicinity of monitoring wells MW25A, MW25B, MW25C, MW28A, MW28B, and MW28C is believed to be a release of chlorinated solvents from the former vapor degreaser area in Building #8. The elevated concentrations of chlorinated VOCs measured in the groundwater samples collected from monitoring wells MW28A, MW28B, and MW28C suggests that separate-phase DNAPLs have migrated offsite within the fractured bedrock aquifer at least 160 feet north of the Bishop Tube property boundary. Based upon the concentrations of chlorinated VOCs detected in the groundwater samples, separate-phase DNAPLs have invaded the fractured bedrock aquifer between monitoring wells MW25 and MW28 to depths ranging from 100 to 280 feet. It should be noted that the findings of the concentration trend analysis (see Section 3.6) show

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that the dissolved-phase and separate-phase (i.e., DNAPL) fractions of the groundwater plume are continuing to migrate within the 100 to 280 foot depth interval of monitoring wells MW25 and MW28.

The source of the DNAPLs occurring in the vicinity of monitoring well MW27B is believed to be a release of chlorinated solvents from the former drum storage area. The elevated concentrations of chlorinated VOCs measured in the groundwater sample collected from monitoring well MW27B suggests that separate-phase DNAPLs have migrated in a southward direction (i.e., hydraulically upgradient) within the fractured bedrock aquifer. The migration of DNAPLs in this direction is believed to have been facilitated by the southeastward dipping foliations contained within the underlying phyllite/schist rock beds (see Figure 10). According to Cohen and Mercer (1993), groundwater concentration trends increasing in a hydraulically upgradient direction is an indication that separate-phase DNAPLs may be present in the subsurface. Such trends are generally caused by the preferential migration of DNAPLs along permeable conduits that exhibit slopes counter to the hydraulic gradient. Based upon the concentrations of chlorinated VOCs detected in the groundwater sample, separate-phase DNAPLs are believed to have invaded the fractured bedrock aquifer in the vicinity of monitoring well MW27B to a depth of 187 feet.

Monitoring well MW26 is situated along a line roughly parallel to the strike of the underlying phyllite/schist rock beds with the former vapor degreaser area in Building #8. The source of the chlorinated solvents occurring in the groundwater of monitoring wells MW26B and MW26C is believed to be principally from a release of DNAPLs in the former vapor degreaser area of Building #8. A supplemental source for the chlorinated solvents contained in the groundwater of monitoring wells MW26B and MW26C may also be from a release of DNAPLs in the former drum storage area. The locations of these two source areas are shown on Figure 12. Based upon the concentrations of chlorinated VOCs detected in the groundwater samples, separate-phase DNAPLs have invaded the fractured bedrock aquifer in the vicinity of monitoring well MW26 to depths ranging from 90 to 230 feet.

The groundwater samples collected from monitoring well MW26C were found to contain the highest concentrations of TCE at the Bishop Tube site. The concentrations of the VOCs detected in the groundwater samples collected from monitoring wells MW26B and MW26C suggest that separate-phase DNAPLs are present in the surrounding bedrock. Based upon the local geologic structure and groundwater flow characteristics, it is believed that DNAPLs released from the former vapor degreaser area in Building #8 have preferentially migrated along the strike of the rock beds to monitoring wells MW26B and MW26C. This supposition is supported by the presence of MTBE in the groundwater at the Bishop Tube site. MTBE is an unleaded gasoline additive first being used in the early 1980's as a replacement for tetra-ethyl lead, and later, in greater concentrations as an oxygenate to produce cleaner burning motor fuels. Historical records suggest that MTBE <u>was</u> <u>not</u> used during the former manufacturing operations at the Bishop Tube site. Based upon its high solubility, MTBE typically migrates at the same rate as the flow of groundwater, behaving as a conservative tracer. The source of the MTBE detected in the groundwater at the Bishop Tube site is attributed to a release of hydrocarbon compounds (i.e., unleaded gasoline) at the neighboring Mobil bulk oil terminal. The presence of MTBE in the groundwater at the Bishop Tube site indicates that this gasoline additive has migrated laterally along the strike of the rock beds. This finding suggests that chlorinated solvents (namely TCE) may also be migrating in a similar fashion in the fractured bedrock aquifer. Monitoring well MW26 is situated near the northeast (i.e., hydraulically downgradient) corner of the Bishop Tube site. Importantly, the separate-phase DNAPLs in the vicinity of monitoring wells MW26B and MW26C may be migrating in an eastward direction beyond the property boundaries of the Bishop Tube site. This supposition is supported by the findings of the concentration trend analysis (see Section 3.6), indicating that the dissolved-phase and

separate-phase (i.e., DNAPL) fractions of the groundwater plume are continuing to migrate beyond the location of monitoring wells MW26B and MW26C. It should be noted that the lateral extent of the dissolved-phase and separate-phase (i.e., DNAPL) fractions of the groundwater plume toward the east has not been determined. It is probable that elevated concentrations of chlorinated solvents along with separate-phase DNAPLs may exist in the fractured bedrock aquifer underlying a portion of the neighboring General Warren Village residential development situated immediately east of the Bishop Tube site.

Elevated concentrations of chlorinated VOCs were detected in the groundwater samples collected from the well at 30 Conestoga Road. The cumulative mole fraction calculations suggest that separate-phase DNAPLs may be present in the bedrock upgradient of this well. The well at 30 Conestoga Road is situated approximately 1,600 feet northeast of the Bishop Tube site. Based upon potentiometric heads and groundwater pathlines generated by the groundwater flow model (see Section 5.3 of the report), the well at 30 Conestoga Road is situated hydraulically downgradient to the Bishop Tube site. As discussed above, separate-phase DNAPLs most likely occur in the fractured bedrock aquifer in the vicinity of monitoring wells MW26 and MW28. The findings of the concentration trend analysis (see Section 3.6) show that the dissolved-phase and separate-phase (i.e., DNAPL) fractions of the groundwater plume are continuing to migrate beyond the multi-screened intervals of monitoring wells MW26 and MW28. The downgradient extent of the separate-phase DNAPL plume within the deeper portions (i.e., +100 feet) of the fractured bedrock aquifer between monitoring wells MW26/MW28 and the well at 30 Conestoga Road has not been determined. Importantly, the concentrations of chlorinated VOCs detected in the groundwater samples suggest that the separate-phase DNAPL plume extends beyond the property boundaries of the Bishop Tube site in a downgradient direction approaching the location of the well at 30 Conestoga Road. The findings of the concentration trend analysis (see Section 3.6) show that the dissolved-phase fraction of the groundwater plume is continuing to migrate beyond the location of the well at 30 Conestoga Road. Importantly, the downgradient extent of the groundwater plume containing chlorinated solvents originating from the Bishop Tube site has not been determined. The reader is referred to Section 5.5.2 of the report regarding an estimate of the groundwater plume length based upon the results of the RT3D groundwater flow model.

An assessment of the stability of a groundwater plume requires information regarding the location of the NAPLs in the subsurface environment and potential saturations within the pore spaces/fractures of the aquifer. This information is particularly important when designing remedial programs for DNAPLs because their high density allows these fluids to sink within an aquifer and migrate along permeable pathways. The results from field studies have shown that partitioning tracers can be used to locate and estimate NAPL saturations in the subsurface (Jin et al., 1995; Nelson and Brusseau, 1996). A recent study published by Davis and others (2003) suggests that naturally occurring radon-222 can be used in lieu of injected partitioning tracers for locating and estimating NAPL saturations in the subsurface environment.

Radon-222 is a naturally occurring radioactive gas that can be present in ground water. Radon-222 concentrations in ground water are continuously produced within an aquifer by the natural decay of uranium-238. The concentrations of radon dissolved within groundwater depend upon the types of rock materials comprising an aquifer and typically vary spatially between different geologic units (Loomis, 1987; Michel and Jordana, 1987).

In groundwater, the background radon concentration ($C_{w,bkg}$) is a function of the radium content (C_{Ra}) in the materials of the aquifer, the radon emanation power (E_p) of the mineral phases comprising the aquifer,

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the bulk density of the aquifer materials (ρ_b), and the porosity (n) of the aquifer materials. These variables can be expressed in the following equation to determine the background radon concentration ($C_{w,bkg}$) (Semprini et al., 2000):

$$C_{w,bkg} = \frac{C_{Ra} E_p \rho_b}{n} \qquad \text{Equation \#7}$$

where:

$C_{w,bkg}$	= Background radon concentration
C_{Ra}	= Radium content in the aquifer materials
E_p	= Radon emanation power of the mineral phases comprising the aquifer
ρ_b	= Bulk density of the aquifer materials
n	= Porosity of the aquifer materials

Radon exhibits the tendency to partition into NAPL from groundwater. Specifically, in the subsurface environment, radon concentrations may be higher in NAPL over that of groundwater due to the effects of partitioning. The linear partitioning coefficient (K) for radon is defined as:

$$K = \frac{C_n}{C_{w,n}} \qquad \text{Equation \#8}$$

where:

K	= Radon linear coefficient
C_n	= Radon concentration in the NAPL phase
$C_{w,n}$	= Radon concentration in the aqueous phase in the presence of NAPL

Both static and push-pull partitioning methods using radon as a tracer can be used to identify and estimate the NAPL saturations in the subsurface environment. The static method involves calculating NAPL saturations from a comparison of radon concentrations in groundwater samples obtained from zones within an aquifer containing free product and intervals where no NAPL is present. This method assumes that equilibrium conditions exist between radon emanation and decay, equilibrium radon partitioning between the water and NAPL phases, and a constant background radon concentration (Semprini et al., 2000). If NAPL is present in the aquifer, radon will be distributed between the water and the NAPL phases as described by the following equation:

$$C_n S_n + C_{w,n} S_w = \frac{C_{Ra} E_p \rho_b}{n} \qquad \text{Equation \#9}$$

where:

S_n	= NAPL saturation in the pore space of the aquifer ($S_n + S_w = 1$)
S_w	= Water saturation in the pore space of the aquifer ($S_n + S_w = 1$)

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$C_{w,n}$ = Radon concentration in the aqueous phase in the presence of NAPL
C_n = Radon concentration in the NAPL phase
C_{Ra} = Radium content in the aquifer materials
E_p = Radon emanation power of the mineral phases comprising the aquifer
ρ_b = Bulk density of the aquifer materials
n = Porosity of the aquifer materials

Assuming linear equilibrium partitioning of radon between the NAPL and water phases, Equation #9 can be rearranged to solve for the NAPL saturation using the following relationship as proposed by Davis and others (2003):

$$S_n = \left(\frac{C_{w,bkg}}{C_{w,n}} - 1 \right) \left(\frac{1}{K-1} \right)$$ Equation #10

where:

S_n = NAPL saturation in the pore spaces of the aquifer
$C_{w,bkg}$ = Background radon concentration outside the zone containing NAPL
$C_{w,n}$ = Radon concentration in the aqueous phase in the presence of NAPL
K = Radon linear coefficient

A published value of K=58 (radon linear coefficient) for radon in TCE was used to compare laboratory and field tests by Davis and others (2002). The value for "K" in TCE was estimated by Cantaloub (2001) by dividing the radon concentration in trichloromethane by the Ostwald coefficient. The Ostwald coefficient is defined as the ratio of the concentration of gas per unit volume in the gas phase (Clever, 1979).

During the October 2003 and February 2004 sampling events, separate groundwater samples were collected from selected monitoring wells at the Bishop Tube site and submitted to the PADEP Bureau of Laboratories, Harrisburg, Pennsylvania for the analysis of radon. The potential NAPL saturations within the fractures of the aquifer surrounding monitoring wells MW02, MW25, MW26C, and MW28C were determined using the radon concentrations ($C_{w,n}$) reported by the PADEP Bureau of Laboratories. It should be noted that problems were encountered in shipping the radon samples to the PADEP Bureau of Laboratories during both the October 2003 and February 2004 sampling events. Specifically, some samples were lost by the courier service, whereas the holding times for other samples were exceeded. Based on this information, the assessment of the potential DNAPL saturations contained within the underlying fractured bedrock aquifer should be considered a rough estimate. The radon concentration measured in the groundwater sample collected from monitoring well MW03 in February 2004 was used for the "background" radon level in the calculations. This value for the "background" radon concentration was chosen based upon the levels of radon measured in the other groundwater samples collected at the site (see Tables 28.1 and 28.2). The potential NAPL saturations (S_n) existing in the fractures of the bedrock surrounding monitoring wells MW02, MW25, MW26C, and MW28C were then calculated using Equation #10 (see Appendix G for calculations).

Based upon the calculations used to determine the cumulative mole fractions in the groundwater samples, separate-phase DNAPLs may be present in the following monitoring wells at the Bishop Tube site: MW02, MW25A, MW25B, MW25C, MW26B, MW26C, MW27B, MW28A, MW28B, MW28C, and the well at 30 Conestoga Road.

The assumption that the radon concentrations would be lower in groundwater samples where separate-phase NAPLs are present in the subsurface is consistent for the groundwater samples collected from monitoring wells MW02, MW25A, MW25B, MW25C, and MW28C. This assumption, however, is inconsistent with the radon concentrations measured in the groundwater samples collected from monitoring wells MW26B and MW28B. The reason for this inconsistency in monitoring wells MW26B and MW28B is unknown, but may be related to lithological differences in the subsurface (creating higher concentrations of radon) at the screened depths of these wells, and/or the measurement/reporting errors by the testing laboratory due to the holding times for some samples being exceeded (see above).

The calculations show that the potential DNAPL saturations in the bedrock materials surrounding monitoring wells MW02, MW25A, MW25B, MW25C, and MW28C range from 0.03% to 40.79%. This later value was calculated using the radon concentrations in the groundwater sample collected from monitoring well MW28C. Although separate-phase DNAPLs have been observed in monitoring well MW28C, a saturation value of 40.79% may be an overestimate of the actual conditions within the fractures of the bedrock surrounding this well.

It should be noted that the calculated NAPL saturations exhibit a decreasing trend with respect to depth within the aquifer surrounding monitoring well MW25. This decreasing trend in the calculated NAPL saturations is believed to be a function of the finite release of chlorinated solvents at the site and the reduction of the fracture porosity with respect to depth in the underlying bedrock. The comparison of the TCE concentrations measured in the groundwater samples to the calculated DNAPL saturations in the fractured bedrock aquifer in the vicinity of monitoring well MW25 shows a close relationship between the fluctuations in these variables (see graph presented in Appendix G). Specifically, as the concentration of TCE increases/decreases within the three screened intervals, the DNAPL saturations show a corresponding increase/decrease. To determine whether or not a linear relationship exists between the TCE concentrations measured in the groundwater samples and the calculated DNAPL saturations, a linear graph comparing these two variables was generated (see Appendix G). This comparison shows a good linear relationship between the TCE concentrations measured in the groundwater samples collected from monitoring wells MW25A, MW25B, and MW25C, and the calculated DNAPL saturations.

These calculated and estimated DNAPL saturation values were used to determine the potential volume of residual DNAPLs contained in the fractured bedrock aquifer surrounding monitoring wells MW02, MW25A, MW25B, MW25C, and MW28C (see Appendix G). These calculations included the determination of the volume of the rock containing residual DNAPLs, a range of porosities for the fractured rock, and an average value for the residual DNAPL saturation. The calculations show that the volume of residual DNAPLs contained in the bedrock surrounding monitoring well MW02 may range from 51 to 103 gallons. Calculations performed for monitoring wells MW25, MW26C, and MW28C show that the following volumes of residual DNAPLs may be present within the fractures of the bedrock surrounding these wells: 4,800 to 9,600 gallons (MW25), 290 to 580 gallons (MW26C), and 1,920 to 3,835 gallons (MW28C). It should be noted that these volumes of residual DNAPL are only a rough estimate, but have been provided to serve as a benchmark for evaluating the feasibility of implementing

different technologies for remediating the chlorinated solvents contained in the fractured bedrock aquifer underlying the project area.

4.0   DISCUSSION OF FATE AND TRANSPORT OF VOC-RELATED CONSTITUENTS

Several VOCs were detected in the groundwater samples collected from the monitoring wells at the Bishop Tube site. The principal constituents of concern in the groundwater are the chlorinated solvents 1,1-DCE, cis-1,2-DCE, trans-1,2-DCE, methylene chloride (dichloromethane), PCE, 1,1,1-TCA, TCE, and vinyl chloride. VOCs exhibit the potential to be mobile in the environment. Their mobility in the vadose zone is a function of low water solubilities and high vapor pressures. In groundwater, the mobility of these compounds is a function of their low absorption to the aquifer materials (i.e., low $[K_{ow}]$ and $[K_{oc}]$ values), low water solubility, and specific gravity (i.e., density) greater than that of water.

In surface media, VOCs will readily volatilize into the atmosphere. Because VOCs are highly mobile in soil, they will leach to underlying groundwater, but will not partition significantly from the water column to soil. In natural water and soil systems, VOCs will slowly biodegrade. Consequently, in subsurface environments, VOCs will tend to persist. Hydrolysis, oxidation, and direct photolysis are not deemed to be important fate processes for the VOCs contained in the groundwater underlying the Bishop Tube site.
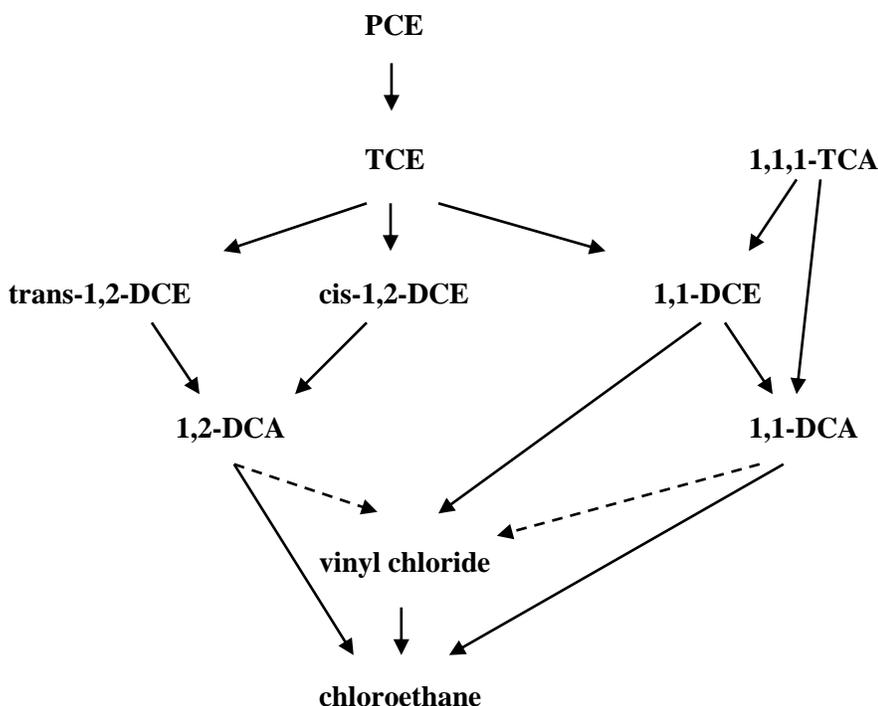
The results of the Supplemental Soil Characterization Baker (2003) found that the soils underlying the former vapor degreaser area #1 (Building #8), the former vapor degreaser area #2 (Building #5), and the former drum storage area contain elevated concentrations of chlorinated solvents (namely TCE). It should be noted that the residual chlorinated solvents and DNAPLs contained in the soils underlying impervious areas at the site (i.e., the former vapor degreaser area #1 in Building #8 and the former vapor degreaser area #2 in Building #5) may be less mobile than those beneath pervious saturated areas (i.e., the former drum storage area). Fluctuations in the concentrations of chlorinated solvents contained in the shallow groundwater flow system underlying the site are believed to be strongly related to temporal amounts of precipitation (i.e., leaching beneath pervious areas) and seasonal saturation of the soils underlying the source areas. Importantly, the impacts to the groundwater quality in the shallow groundwater flow system may be reduced by the remediation of the soils underlying the former vapor degreaser area #1 (Building #8), the former vapor degreaser area #2 (Building #5), and the former drum storage area that contain residual amounts of chlorinated solvents and DNAPLs.

Chlorinated solvents (including 1,1-DCE, cis-1,2-DCE, trans-1,2-DCE, methylene chloride, PCE, 1,1,1-TCA, TCE, and vinyl chloride) tend to persist in the environment because they are recalcitrant to both chemical and biological degradation. Information published by Barbee (1994) suggests that the average half-life for abiotic (i.e., chemical) transformations of chlorinated solvents ranges from two months to greater than one million years. Of the chlorinated solvents, the ethene compounds tend to be the most resistant to abiotic degradation. Anaerobic reductive dehalogenation is recognized as the primary biodegradation process in groundwater. Aerobic biodegradation, however, may be locally important in some shallow groundwater flow systems.

The rate and extent of abiotic (i.e., chemical) and biotic (i.e., microbiological) transformations of chlorinated solvents is a function of the physiochemical properties in the subsurface, the soil and groundwater chemistry, and the types of the indigenous microbial populations. The molecular size, structure, types of chemical bonds, aqueous solubility, and concentration of the chlorinated solvents collectively regulate the physiochemical transformations in the subsurface. Soil and groundwater

chemistry, such as pH and redox (Eh) conditions together with the type and availability of electron acceptors/donors also greatly control the degradation rate of chlorinated solvents in the subsurface. Biotransformation of chlorinated solvents is also regulated by the microbe's accessibility to the electron acceptors/donors, other constituents (i.e., heavy metals) that may create toxic conditions, the capacity of the indigenous microbial community to metabolize or co-metabolize the parent and daughter compounds, and the availability of essential microbial growth factors (i.e., temperature, pH, and nutrients).

Chlorinated solvents exist in a formally oxidized state due to highly electronegative halogen substitutes on the molecules. Accordingly, the parent compounds are generally more susceptible to reduction than to oxidation reactions (Sims et al., 1991). With increasing halogenation, reduction becomes much more likely than oxidation (Vogel et al., 1987). According to the summary by Kollig and others (1990) and McCarty (1991), the primary and secondary anaerobic abiotic transformation pathways for the types of chlorinated solvents present in the groundwater at the Bishop Tube site would be:

PCE

↓

TCE   **1,1,1-TCA**

trans-1,2-DCE   **cis-1,2-DCE**   **1,1-DCE**

1,2-DCA   **1,1-DCA**

vinyl chloride

↓

chloroethane

Based on this transformation pathway, vinyl chloride may or may not originate as a daughter product from the breakdown of more halogenated chlorinated solvents. It should be noted that cis-1,2-DCE is considered an ethene-based organic compound, and is considered to be one of the most non-reactive chlorinated solvents (Barbee, 1994).

Reductive dehalogenation is believed to be the most important process for biotransforming chlorinated solvents in the subsurface environment. This reaction is carried out primarily by indigenous populations of methane-oxidizing bacteria (i.e., methanotrophs) occurring in the soils and groundwater. During reductive dehalogenation, methanotrophs do not directly use chlorinated solvents as an energy source, but biodegrade the compounds by co-metabolism (McCarty, 1991). In co-metabolic reductive dehalogenation, bacterial growth is supported by a primary substrate that supplies energy and carbon,

causing the release of enzymes and cofactors. These enzymes and cofactors serve as a secondary substrate, facilitating the biodegradation of the chlorinated solvents. The biodegradation of some chlorinated solvents may produce daughter products that are more recalcitrant and of greater health threat than the parent compound. An example reaction would be: PCE→TCE→cis-1,2-DCE→vinyl chloride (chloroethene). Moreover, this process may create less halogenated daughter products that exhibit lower reaction rates under anaerobic conditions (McCarty, 1991). Several daughter products (i.e., 1,1-DCA, 1,2-DCA, 1,1-DCE, cis-1,2-DCE, trans-1,2-DCE, vinyl chloride [VC], and chloroethane) were detected in the monitoring wells at the Bishop Tube site. This suggests that some natural degradation of 1,1,1-TCA, PCE, and TCE is occurring in the groundwater underlying the study area.

Based upon the information outlined above, cis-1,2-DCE, trans-1,2-DCE, 1,1-DCE 1,1-DCA and 1,2-DCA, may be daughter products from the breakdown of PCE and TCE in groundwater at the Bishop Tube site. The chlorinated solvents 1,1-DCE and 1,1-DCA may also be related to the breakdown of 1,1,1-TCA. Vinyl chloride and chloroethane may be daughter products of either the breakdown of PCE/TCE or 1,1,1-TCA. It appears that the abiotic and biotic transformations of PCE→TCE→cis-1,2-DCE→1,2-DCA→VC→chloroethane and 1,1,1-TCA→1,1-DCE→1,1-DCA→VC→chloroethane are occurring at only relatively slow rates within the fractured bedrock aquifer in the vicinity the Bishop Tube site. The cause(s) for the reduced rate of the transformations is unknown, but may be related to the aerobic conditions existing in the fractured bedrock aquifer. Information published by Barbee (1994) suggests that low rates of biodegradation of chlorinated solvents may be related to the lack of sufficient substrate for microbial growth. In addition, the rate of biodegradation may also be hindered by the relatively high concentrations of TCE dissolved in the groundwater. Higher rates of biodegradation have been reported by Barbee (1994) along the edges of plumes where the concentrations of solvents are diluted.

This information combined with the water chemistry data at the Bishop Tube site suggests that the existing conditions in the fractured bedrock aquifer may not be conducive for the rapid breakdown of PCE→TCE→cis-1,2-DCE→1,2-DCA→VC→chloroethane or 1,1,1-TCA→1,1-DCE→1,1-DCA→VC→chloroethane It is expected that the VOCs found in the groundwater at the Bishop Tube site will only slowly biodegrade over time. Remedial measures, by adding supplementary amendments to stimulate abiotic and biotic transformations, however, would be required if more rapid degradation of the VOCs in the groundwater is deemed necessary.

5.0   GROUNDWATER FLOW MODEL

A discussion of how the three-dimensional groundwater flow model was developed and the modeling results are discussed in the following sections.

5.1   <u>Summary of Previous Groundwater Flow Models Developed for the Study Area</u>

The most notable groundwater flow model developed for the Chester Valley area was published by Sloto (1990). Sloto created a single-layer, two-dimensional MODFLOW simulation of the Valley Creek Basin. The Valley Creek Basin is eight miles in length, extending from East Whiteland (western edge) to Valley Forge (eastern border). The basin is about three miles in width (north to south). The model domain developed by Sloto (1990) was actually much larger than the Valley Creek Basin, encompassing an area 18 miles long and 4 miles wide from Brandywine Creek in Downingtown eastward through the carbonate-rich Chester Valley to the Schuylkill River in western Montgomery County (Figure 13).

The MODFLOW model developed by Sloto (1990) recognized that the rocks underlying the Chester Valley area were heterogeneous, with hydraulic conductivity varying within each rock type. Based upon this heterogeneity, each lithology or similar lithologies were assigned a distinctive value for K_x (i.e., hydraulic conductivity in the x-direction). The model also incorporated anisotropy (preferred groundwater flow direction) along the strike of the bedding planes (i.e., parallel to the overall valley trend of ~N72ºE). The 5:1 anisotropy (i.e., K_x and K_y directions) in hydraulic conductivity along strike (estimated by Sloto using MODFLOW) resulted largely from the geologic structure of the valley, in which the beds are dipping very steeply (80º toward the SE, Sloto, 1987). Not only are the carbonate beds dipping, but they are also fractured and many of the bedding planes and fractures have been enlarged by dissolution. All of these factors constrain groundwater to flow parallel to the bedding strike.

The objectives of the groundwater flow model developed by Sloto (1990) included: 1) simulating the average water budget in the basin; and 2) estimating the effects of increased well and quarry pumping on baseflow and underflow. Sloto (1990) calibrated his groundwater flow model to simulate the observed (both measured and reported) heads in observation wells throughout the Valley Creek Basin and the base flow measured in local streams within the basin in 1984. The model report concluded with some predictive simulations regarding projected groundwater supply needs for the Chester Valley area.

The model developed by Sloto (1990) provided the basis for the current modeling effort, which was modified to include four layers and contaminant transport from the Bishop Tube site.

5.2 Setup of the Groundwater Flow Model

The numerical groundwater model input data for MODFLOW-2000 were processed using the Groundwater Modeling System (GMS version 4.0) authored by the Environmental Modeling Research Laboratory (EMRL, 2003) at Brigham Young University. GMS is a graphical pre- and post-processor that allows the user to construct, edit, and print the finite-difference grid size and spacing, the hydrogeologic input parameters, and the output from the groundwater model. MODFLOW-2000 is the numerical finite-difference groundwater flow code developed by the U.S. Geological Survey (Harbaugh et al., 2000).

MODFLOW-2000 was used to simulate the existing groundwater flow regime in the area surrounding the Bishop Tube Site. The model was calibrated to groundwater elevations measured and reported for observation wells within Chester Valley by iteratively varying the boundary conditions, the recharge values, and/or the hydraulic conductivity of the lithologies until a statistical fit was made to the groundwater targets for the site (based on existing groundwater elevation data).

5.2.1 Groundwater Elevation Data

The groundwater elevation (head) calibration targets included historical data (Sloto, 1989 and 1990) and site-specific data collected from the monitoring wells at the Bishop Tube site during the period from 2001 to 2003. The site-specific data, including means and standard deviations, used to calibrate the model near the Bishop Tube site is listed in Table 29. The head data from Sloto (1990) were used to calibrate the other portions of the model domain.

5.2.2   Model Grid and Boundary Conditions

The location of the model grid is shown in Figure 3. The entire model grid and the designated boundary conditions (i.e., rivers and streams are blue) in the uppermost model layer (layer 1) are displayed on Figure 14. The model was set up using a denser grid line spacing near the Bishop Tube site to improve accuracy. The areas within the grid frame that have no cells are designated as being inactive. The grid has four layers and the surface of the top layer is coincident with the land surface (Figure 15). Each underlying layer is 100 feet thick. This was done so that hydraulic conductivity could be varied with respect to depth (see Section 5.2.4).

The boundary conditions near the Bishop Tube site are shown in Figure 16. The western boundary of the model domain is made of MODFLOW river cells representing Brandywine Creek (near Downingtown). The eastern boundary of the model domain is also made of river cells representing the Schuylkill River (across from Norristown). The northern and southern boundaries of the model domain are no-flow boundaries because they represent drainage divides at topographic highs. MODFLOW river cells and drain cells were used only in the uppermost layer (layer 1) to simulate baseflow discharge to the streams throughout the model domain. River cells can always contain water at a given elevation and can contribute water to the system (i.e., losing stream) if conditions allow. Drain cells (that do not contribute water to the system) were used for the smaller tributaries in the uplands where the streams are typically ephemeral. Elevations of the streams were taken from five USGS topographic maps of the area (Downingtown, Malvern, Norristown, Unionville, and Valley Forge). The surface elevations of the streams obtained from these maps are believed to be within one or two feet of the actual stream elevations.

5.2.3   Recharge

Chester County, Pennsylvania receives on average about 47.25 inches/year (1.08E-02 ft/d) of annual rainfall. Several factors influence the infiltration of precipitation, including evapotranspiration, surface conditions, topography, soil type, and water content of the vadose (unsaturated) zone. Sloto (1990) estimated an average infiltration of 24.84 inches/year (5.67E-03 ft/d) or about 52% of annual precipitation for the carbonate rocks and about half that amount, 12.42 inches/year (2.83E—03 ft/d) or 26% of annual precipitation for the non-carbonate rocks.

For the development of the Bishop Tube groundwater model, these recharge values were increased by 25% during the calibration process (31.05 inches/year for carbonates and 15.53 inches/year for non-carbonates) with the addition of an intermediate recharge value for the Triassic sediments (26.25 inches/year). The distribution of recharge within the model domain is displayed in Figure 17. The distribution of recharge (in ft/day) was designated only to layer one (the uppermost layer).

5.2.4   Hydraulic Conductivity

The mean hydraulic conductivity values used in the Bishop Tube flow model were derived from the groundwater flow model developed by Sloto (1990). The K values used in the model developed by Sloto (1990) were estimated on a large scale by calibrating the model to stream flow data and measured (and reported) groundwater elevation (head) values in the Valley Creek Basin. As shown in the center column on Table 29, Sloto (1990) represented each hydrogeologic unit, (comprised of one or more geologic formations) with a single hydraulic conductivity value in the x-direction (K_x). Because of the anisotropy

created by the regional geologic structure (discussed above), the K_y values were always less than the K_x values by a factor of five.

The hydrogeology of the area is shown on Figure 18. GMS uses "materials" to distribute hydraulic conductivity to MODFLOW. In this case, the material types are synonymous with the hydrogeologic units. The material distribution is assumed identical in all four layers because the beds are steeply dipping (Figure 15).

The right column of Table 29 represents the final calibrated value of the *geometric mean* of K_x in all four layers of the current model. The actual K_x values used for each MODFLOW layer, however, were different than the mean value shown in Table 29. The actual values used in the model were calculated based on the assumption that the K_x values reported by Sloto (1990) represented the geometric mean. The theory used to develop the Bishop Tube model was that hydraulic conductivity (due to secondary porosity) is directly proportional to fracture frequency versus depth below ground surface. Sloto (1990) found that the fracture frequency diminishes with depth (see Figure 19). Accordingly, the development of the Bishop Tube model attempted to use this relationship to reduce the hydraulic conductivity values with respect to depth. This approach is reasonable since the hydraulic conductivity of the rocks underlying Chester Valley is related mostly to the development of secondary porosity. The K_x values estimated by Sloto (1990) were assumed to represent the geometric means of the entire unit thickness. Using each mean K_x value and the ratio of fracture frequency versus depth, factors to reduce the K_x values for each of the four MODFLOW layers were calculated (see Table 30).

The center column of Table 29 shows that Sloto (1990) estimated a K_x value of 10 ft/day (3.5E-03 cm/sec) for the Conestoga formation, which he adjusted downward during calibration from his initial estimate of 32 ft/day (1.1E-02 cm/sec). The final calibration for the current modeling effort resulted in a geometric mean K_x value of 3.0 ft/day for the Conestoga formation. The difference between this effort and the calibrated K_x value by Sloto (1990) may be due to differences in the model layer thicknesses. The Bishop Tube model is four times thicker than the model developed by Sloto (1990) (i.e., 400 feet versus 100 feet). It should be noted that the overall transmissivity (T) of the Bishop Tube model [3 ft/day (mean) x 400 ft = 1,200 ft^2/day] is approximately equal to the value of T (10 ft/day x 100 ft = 1,000 ft^2/day) used by Sloto (1990).

Sloto estimated a K_x value of 5.0 ft/day (1.8E-03 cm/sec) for the Octoraro Phyllite. The final calibration for the current modeling effort resulted in a K_x value of 0.5 ft/day for the Octoraro Phyllite. As discussed above, the difference between this effort and the calibrated K_x value by Sloto (1990) may be due to differences in the model layer thicknesses. The Bishop Tube model is four times thicker than the model developed by Sloto (1990) (i.e., 400 feet versus 100 feet). The overall transmissivity of the current model is roughly similar (0.5 ft/day x 400 ft = 200 ft^2/day) to the value of T (5 ft/day x 100 ft = 500 ft^2/day) used by Sloto (1990). The K_x values of the other units listed in Table 29 were kept the same as in the model developed by Sloto (1990).

A pumping test performed at the Bishop Tube site in 2002 (Baker 2002b) determined a hydraulic conductivity value of 8.3 ft/day for the fractured bedrock of the Conestoga formation (i.e., based upon the recovery results of bedrock wells MW09 and MW18). The pumping test results for the other observation wells were not considered for use in the Bishop Tube model because: 1) these wells were shallower than the pumping well; and 2) these wells monitored water level fluctuations in the soils (overburden) rather than the bedrock.

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5.3   MODFLOW Results

The simulated steady-state water table contours for the overburden groundwater at the Bishop Tube site are shown in Figures 20 and 21. The simulated groundwater elevation contours from layer 1 (0-100 feet below ground surface [bgs]) superimposed upon the hydrogeology (MODFLOW materials) near the Bishop Tube site is shown on Figure 20. The groundwater elevation contours in layers 2, 3, and 4 are very similar to those in layer 1, therefore; the contours in the underlying three layers are not shown. The same groundwater elevation contours superimposed upon a USGS topographic quadrangle map (Malvern) are depicted on Figure 21. The simulated groundwater contours in the other three layers are very similar to those in layer 1.

The simulated groundwater contours closer to the Bishop Tube site, as superimposed upon an aerial photograph, is shown on Figure 22. The calibration targets for the Sloto data (1990) in the area near the Bishop Tube site are displayed with the simulated groundwater contours on Figure 23. There are many points in the Sloto (1990) data that must be used qualitatively because they were collected by measurement or reported by the well owner in 1983. Many changes have occurred in Chester Valley since these water levels were recorded. For example, the water elevation in the private well at 54 Conestoga Road is reported to be more than 30 feet lower than the adjacent stream elevation. This water level must have been measured when the well was pumping. As such, this point should be deleted from the calibration data set. Other elevations are reported to be much higher or lower than the simulated heads and it is impossible to tell if these changes are due to the operation of the quarries or landfills within the valley over the past 20 years. Many such potential inaccuracies exist in the Sloto (1990) data set, but it was still useful as a starting point for calibration of the model domain.

"Error" at a particular calibration target location (well) is the difference between the observed and the measured value of groundwater elevation (head). The overall error of the model is measured by the root mean square (RMS) error statistic. The RMS error is calculated by the equation:

$$RMS = \left[\sum \frac{(h_c - h_o)^2}{n} \right]^{\frac{1}{2}} \qquad \text{Equation \#11}$$

where:

h_c = computed head at a single well
h_o = observed or reported head at a single well
n = number of wells

Graphical comparisons and statistical summaries of the observed versus the simulated heads for the head calibration targets throughout the Valley Creek Basin (Sloto, 1990) are displayed in Figure 24. While there are no established rules regarding specific RMS error values to judge the adequate degree of calibration of a particular model, the objective is to minimize error. An RMS error value less than 10% of the change in elevation across the subject area is generally considered as being sufficiently precise. The RMS error for the Valley Creek basin in the Bishop Tube model is 39.2 feet. This value is less than 50 feet, which is 10% of the difference between the maximum and minimum observed heads (525 – 25 = 500 feet) across the model domain (see Figure 24 for maximum and minimum values).

The simulated groundwater contours immediately downgradient of the Bishop Tube site are shown on Figure 25. The calibration targets for the site-specific data in the area near the Bishop Tube site are displayed with the simulated groundwater contours on Figure 26. An extreme close-up of the Bishop Tube site with the onsite calibration targets is displayed on Figure 27. A graph showing the computed versus observed heads for the site-specific data is presented in Figure 28. The maximum head difference in the site-specific data set is about 90 feet (410 - 320 = 90 feet) from the upgradient well (MW01) to the downgradient monitoring wells along Conestoga Road. The calculated RMS error for this model near the Bishop Tube site was 4.79 feet, which is less than 9.0 feet (= 10% of 90 feet).

A MODPATH pathline analysis was run using the calibrated, steady state MODFLOW solution. The groundwater pathlines generated by MODPATH are shown on Figure 29. Particles were started in all four layers around the footprint of the Bishop Tube site to estimate the potential for dense non-aqueous phase liquid (DNAPL) migration away from the facility. The particles were allowed to flow downgradient for 52 years, covering the operational period of the facility from approximately 1951 to 2003. The particles in layer 1 discharged to Little Valley Creek within about 2,100 feet of the site within about five years of release (time interval between red arrows is two years). MODPATH particles in layers 2 and 3 took longer to reach Little Valley Creek, while those in layer 4 did not reach the creek in 52 years.

The groundwater pathlines generated by MODPATH in layer 2 are shown on Figure 30. Based upon the rate of groundwater flow, the particles in layer 2 discharged to Little Valley Creek (within 4,000 feet of the Bishop Tube site) in about 20 years (time interval between red arrows is five years). The model shows that particles in layer 3 (see Figure 31) reached Little Valley Creek (within about 5,500 feet of the Bishop Tube site) in 40 to 50 years (time interval between red arrows is 20 years).

Based upon the rate of groundwater flow, the particles in layer 4 (see Figure 32) have not yet reached Little Valley Creek within the allotted time (52 years). The particles in layer 4 have migrated about 3,200 feet from the Bishop Tube site and will continue to migrate at depth parallel to the creek until ultimately discharging to surface water in the future (time interval between red arrows is 20 years).

A cross-sectional profile of particle movement parallel to Little Valley Creek in all four layers (time interval between red arrows is 20 years) is shown in Figure 33. It should be noted that the lack of red arrows for particles in originating in layers 1 and 2 indicates that groundwater has discharged to Little Valley Creek in less than 20 years.

The groundwater model shows that there is a high potential for the offsite migration of chlorinated solvents from the Bishop Tube site. Importantly, the results of the groundwater model show that the chlorinated solvents contained in the groundwater plume may pose a risk to human and ecological receptors situated downgradient of the site.

It should be noted that representatives of the PADEP Southeastern Regional Office conducted a detailed surface water study of Little Valley Creek (approximately 1.5 miles downstream from the Bishop Tube site) in May 2003. This surface water study included the collection and analysis of 24 discrete surface water samples from Little Valley Creek, un-named tributary streams, and springs. The analytical results showed that the surface water samples collected from Little Valley Creek adjacent to the Bishop Tube site contained elevated concentrations (SW03 at 55 μg/l) of TCE. The concentrations of TCE detected in the surface water samples collected from Little Valley Creek were found to gradually decrease between the

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Bishop Tube site and sampling point SW09 (located approximately 800 feet downstream of Conestoga Road or 4,500 feet northeast of the Bishop Tube site). The surface water sample collected at sampling point SW09 was found to contain up to 9.5 µg/l of TCE, which is much higher than the concentration of TCE measured in the surface water sample collected from SW08 near the bridge over Little Valley Creek at Conestoga Road (SW08 was collected approximately 800 feet upstream of SW09). The groundwater modeling results show that the pathlines developed using MODFLOW and MODPATH parallel Little Valley Creek northeast of the Bishop Tube site, ultimately reaching the stream approximately 600 to 800 feet downstream of Conestoga Road (i.e., near surface water sampling point SW09). The decrease in TCE in Little Valley Creek between the Bishop Tube site and surface water sampling point SW09 is attributed to dilution effects created by the discharge of groundwater via baseflow containing only low or no chlorinated solvents. These findings support the results of the groundwater model, showing that the groundwater pathlines most likely parallel Little Valley Creek northeast of the Bishop Tube site, ultimately discharging groundwater via baseflow to the stream approximately 600 to 800 feet downstream of Conestoga Road (see Figures 30 and 31). Importantly, impacts to the water quality of Little Valley Creek (an "*Exceptional Value*" stream) and the indigenous fauna and flora may be greatest immediately adjacent to the Bishop Tube site and along the more distant downstream section of the stream where deep groundwater originating from the Bishop Tube facility is discharging via baseflow.

5.4   Sensitivity Analysis

In order to determine the sensitivity of the calibrated flow model, the values of selected input parameters were varied across the entire domain and the model was re-run for each change. The selected sensitivity parameters included changes in: horizontal hydraulic conductivity (K_H), vertical hydraulic conductivity (K_V), vertical anisotropy (K_H/K_V), horizontal anisotropy (K_y/K_x), and recharge.

A graph comparing the above referenced sensitivity parameters is shown on Figure 34. Based on the local site-specific groundwater elevation (head) data, the calibrated model is sensitive to the following changes (in decreasing order):

1.   Increasing recharge – most sensitive parameter
2.   Increasing K_H
3.   Increasing horizontal anisotropy (K_y/K_x)
4.   Decreasing K_H
5.   Decreasing recharge
6.   Decreasing horizontal anisotropy (K_y/K_x) – least sensitive parameter
7.   Increasing or decreasing vertical anisotropy – least sensitive parameter

Based on local, site-specific data, the model is relatively insensitive to any change in vertical anisotropy (K_H/K_V).

A graph comparing the above referenced sensitivity parameters to the regional head data from Sloto (1989 and 1990) is shown on Figure 35. This comparison shows that only the following three parameters significantly affected the calibrated model results:

1.   Increasing recharge – most sensitive parameter
2.   Decreasing K_H

3. Decreasing horizontal anisotropy (K_y/K_x)

Based on regional data from Sloto (1989 and 1990), the model is relatively insensitive to changes in all other parameters.

This sensitivity analysis also shows that each of the curves are at their minimum values near the calibrated input values (multiplication factor = 1); therefore, further calibration of the model with the current data sets (local and regional) would not likely result in significant improvements in the RMS error value. Unless the quality of the calibration data sets are improved (i.e., collecting a comprehensive, synoptic data set over the entire region), a much more accurate model is not possible at this time.

5.5 RT3D Contaminant Transport and Bioremediation Model

The reactive transport model RT3D (Clement, 1997, 1999) was used to simulate the fate of TCE and its degradation products, cis-1,2-dichloroethene (DCE) and vinyl chloride (VC). The input parameters used in the RT3D model for the Bishop Tube Site are listed in Table 31. A detailed discussion of these input parameters are presented in Baker (2002b).

The RT3D model simulation was conducted for a time period of 52 years from 1951 to 2003, covering the operational period of the facility. The MODFLOW-2000 steady-state groundwater flow solution described in Section 5.3 was used to run the RT3D model simulation. RT3D was run assuming the following release scenario:

1951: Plant becomes operational processing precious metals using chlorinated solvents (TCE) in a relatively small area of the facility (i.e., Building 5).
1967: Property is sold. Facility and processes were changed for the manufacture of stainless steel tubing. A vapor degreaser and distillery units (used to recycle spent solvents - TCE) were installed and used in a larger portion of the facility (Building 8). Solvent product (DNAPL) release assumed to have started at this time. DNAPL starts to sink into lower layers.
1983: Assumed date that solvent product release ceases from the surface; however, DNAPL continues to sink into lower layers.
2003: Present-day end of scenario.

Because of the limited amount of deep offsite monitoring wells, only a general comparison can be made of the simulated results to the actual solvent concentrations in groundwater.

5.5.1 TCE Source Location and Strength

Previous investigations have identified three major source areas of chlorinated solvents at the Bishop Tube site (Baker, 2002a, 2002b, 2003). The locations of these source areas are shown on Figure 12. These three source areas lie within the footprint of the site property. The location of the simulated TCE source area for the RT3D model is shown on Figure 36. Some lateral spreading of TCE was assumed to have occurred from the surface to the water table such that the only one TCE plume was generated and plumes from individual sources were not discernible.

Graphs showing the simulated TCE concentrations released over time into the groundwater from the source area since 1951 are displayed in Figure 37. Each graph shows one of the three uppermost model

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layers through which DNAPLs are believed to have migrated. As a worst-case assumption, the strengths (i.e., dissolved concentrations) of the simulated TCE sources over time were assumed to equal the laboratory solubility limit (1,100 mg/L) for this compound.

The operational conditions between 1951 and 1967 are depicted on Figure 37a. During this time period the TCE sources are assumed active, but limited to relatively minor concentrations. Starting in mid-1967, major releases of TCE were assumed to begin at the site based upon the reported operating history of the facility. This included the release of TCE from the former vapor degreaser/distillery located in Building #8 and the former drum storage area.

By 1983, the release of TCE from the former vapor degreaser/distillery and the former drum storage area are assumed to diminish, corresponding to the time when the federal RCRA regulations were being implemented/enforced (see Figure 37b). After 1983, the DNAPL continued to sink into the subsurface (see Figure 37c).

5.5.2 RT3D Results

The results of the RT3D model simulation for the migration of TCE, cis-1,2-DCE, and vinyl chloride in the fractured bedrock aquifer are discussed in the following sections.

5.5.2.1 Trichloroethylene

The simulated TCE plume in the uppermost layer (layer 1) after two years (i.e., through December 1952) of manufacturing operations at the Bishop Tube site is shown on Figure 38. After 16 years of operation (i.e., through 1967), the RT3D model predicted that detectable levels of TCE (>0.001 mg/L) had migrated about 1,900 feet (0.36 miles or 0.58 km) from the facility (see Figure 39). By 1967, the RT3D model predicted that TCE concentrations equaling 100 mg/L had migrated about 1,000 feet (0.19 miles or 0.3 km) from the Bishop Tube facility.

Starting in mid-1967, it was assumed that major releases of solvent began from one or more individual sources (i.e., the former vapor degreaser in Building #8 and the former drum storage area) at the site. The resulting increase in TCE concentrations about 18 months later (December 1968) is depicted in Figure 40. This solvent release was assumed to have continued through 1983. It should be noted that the residual slug(s) of TCE DNAPL were assumed to be moving vertically in the subsurface from mid-1967 to the present (January 2003).

The simulated TCE plume in January 2003, 52 years after the plant opened is depicted in Figure 41. The model predicted that detectable concentrations of TCE (>0.001 mg/L) have migrated almost 3,000 feet from the site (0.55 mi or 0.9 km). The maximum simulated TCE concentration in layer 1 was about 280 mg/L located about 100 feet downgradient of the facility.

It should be noted that the RT3D model shows some TCE concentrations at locations upgradient of the source area. This artifact is the result of numerical dispersion in the RT3D model due to the relatively large cell size. A further discussion of the limitations of the RT3D model is included in Section 5.6.

The following paragraphs discuss only the end-result of the RT3D model simulation for TCE and its degradation products in layers 2, 3, and 4. A detailed discussion for the formation of the plumes in layers 2, 3, and 4 is not provided because they are all similar to that described for layer 1 above.

The lateral extent for the migration of the simulated TCE concentrations in layer 2 over the last 52 years is shown in Figure 42. Detectable TCE concentrations of TCE were predicted to have migrated about 2,700 feet (0.51 miles or 0.82 km) downgradient of the Bishop Tube site. The maximum simulated TCE concentration in layer 2 was about 1,100 mg/L located directly beneath the facility (representing DNAPL at the theoretical solubility limit of TCE).

The lateral extent for the migration of the simulated TCE concentrations in layer 3 over the last 52 years is shown in Figure 43. Detectable TCE concentrations were predicted to have migrated about 2,300 feet (0.44 miles or 0.7 km) feet from the Bishop Tube site. The maximum simulated TCE concentration in layer 3 was about 780 mg/L located directly beneath the facility.

The lateral extent for the migration of the simulated TCE concentrations in layer 4 over the last 52 years is shown in Figure 44. Detectable TCE concentrations were predicted to have migrated about 1,800 feet (0.34 miles or 0.55 km) from the Bishop Tube site. The maximum simulated TCE concentration in layer 4 was about 40 mg/L located directly beneath the facility.

5.5.2.2 cis-1,2-Dichloroethene

The lateral extent for the migration of the simulated cis-1,2-DCE concentrations in layer 1 over the last 52 years is shown in Figure 45. The RT3D model predicted that detectable concentrations (>0.001 mg/L) of cis-1,2-DCE have migrated almost 3,200 feet from the site (0.61 mi or 0.98 km). The maximum simulated cis-1,2-DCE concentration in layer 1 was 4.4 mg/L located about 1,100 feet (0.21 miles or 0.34 km) downgradient of the facility.

The lateral extent for the migration of the simulated cis-1,2-DCE concentrations in layer 2 over the last 52 years is shown in Figure 46. Detectable concentrations of cis-1,2-DCE were predicted to have migrated about 3,200 feet (0.61 miles or 0.98 km) from the Bishop Tube site. The maximum simulated cis-1,2-DCE concentration in layer 2 was 17.3 mg/L located about 700 feet downgradient of the facility.

The lateral extent for the migration of the simulated cis-1,2-DCE concentrations in layer 3 over the last 52 years is shown in Figure 47. Detectable concentrations of cis-1,2-DCE were predicted to have migrated about 2,500 feet (0.47 miles or 0.76 km) from the Bishop Tube site. The maximum simulated cis-1,2-DCE concentration in layer 3 was 9.2 mg/L located about 300 feet downgradient of the facility.

The lateral extent for the migration of the simulated cis-1,2-DCE concentrations in layer 4 over the last 52 years is shown in Figure 48. Detectable concentrations of cis-1,2-DCE were predicted to have migrated about 1,900 feet (0.36 miles or 0.58 km) from the Bishop Tube site. The maximum simulated cis-1,2-DCE concentration in layer 4 was 1.0 mg/L about 100 feet downgradient of the facility.

5.5.2.3 Vinyl Chloride

The lateral extent for the migration of the simulated vinyl chloride concentrations in layer 1 over the last 52 years is shown in Figure 49. The RT3D model predicted that detectable concentrations (>0.001 mg/L)

of vinyl chloride have migrated almost 3,400 feet (0.64 mi = 1.0 km) from the Bishop Tube site. The maximum simulated vinyl chloride concentration in layer 1 was 0.046 mg/L located about 1,000 feet (0.19 miles or 0.3 km) downgradient of the facility.

The lateral extent for the migration of the simulated vinyl chloride concentrations in layer 2 over the last 52 years is shown in Figure 50. Detectable concentrations of vinyl chloride were predicted to have migrated about 3,500 feet (0.66 miles of 1.07 km) from the Bishop Tube site. The maximum simulated vinyl chloride concentration in layer 2 was 0.190 mg/L located about 1,000 feet (0.19 miles or 0.3 km) downgradient of the facility.

The lateral extent for the migration of the simulated vinyl chloride concentrations in layer 3 over the last 52 years is shown in Figure 51. Detectable concentrations of vinyl chloride were predicted to have migrated about 2,600 feet (0.49 miles or 0.79 km) from the Bishop Tube site. The maximum simulated vinyl chloride concentration in layer 3 was 0.115 mg/L located about 600 feet downgradient of the facility.

The lateral extent for the migration of the simulated vinyl chloride concentrations in layer 4 over the last 52 years is shown in Figure 52. Detectable concentrations of vinyl chloride were predicted to have migrated about 1,700 feet (0.32 miles or 0.52 km) from the Bishop Tube site. The maximum simulated vinyl chloride concentration in layer 4 was 0.017 mg/L about 350 feet downgradient of the facility.

5.6   Discussion of RT3D Results

As mentioned above, a comparison of the simulated RT3D results with observed concentrations can only be made in a generalized way due to sparse number of deep offsite monitoring wells. In addition, the locations of some monitoring wells (i.e., 54 Conestoga Road) are only known, spatially, to within several hundred feet.

Graphs comparing the RT3D simulated (i.e., computed) versus the observed concentrations of TCE and cis-1,2-DCE are displayed in Figures 53 and 54, respectively. These graphical plots show that the RT3D model simulation consistently overestimates the concentrations of TCE and cis-1,2-DCE, except for the maximum observed values, which are reasonably close. This artifact of the RT3D model is likely due to the relatively large cell size used near the source area.

The use of large cell sizes assumes that conditions are the same over wide areas. Such an assumption is suitable for modeling groundwater flow conditions. It should be noted that the RT3D model required the same cell sizes as those used for the MODFLOW groundwater flow model. The selection of cell sizes for the MODFLOW groundwater model was based upon the objective to develop a usable three dimensional model for simulating "regional" groundwater flow through the carbonate rocks underlying Chester Valley, respecting the time and budget constraints of the project. The chlorinated solvent concentrations near the site, however, are changing rapidly over relatively short distances. It is recognized that the large cell sizes used for the RT3D model overestimate the concentrations of TCE and cis-1,2-DCE, albeit at the maximum observed values. Accordingly, future refinements to the MODFLOW and RT3D models should incorporate the use of smaller cell sizes to obtain a more accurate solution where concentrations are fluctuating rapidly in the area near the Bishop Tube site. Moreover, the accuracy of the model can be improved further if the offsite database is expanded.

A graph comparing the RT3D simulated (i.e., computed) versus the observed concentrations of vinyl chloride is displayed in Figure 55. This graphical plot shows that the RT3D model simulation for vinyl chloride is more accurate than the simulations for TCE and cis-1,2-DCE. The improved accuracy is likely due to the lower concentrations of vinyl chloride over the area.

The RT3D model results indicate that, even with the caveat described above, the potential exists for the migration of significant amounts of TCE and its degradation products from the Bishop Tube site. The principal area of concern is situated approximately 3,500 feet (0.7 miles or 1.1 km) northeast of the Bishop Tube site, in a line sub-parallel to Little Valley Creek. The RT3D model suggests that the width of the affected area is approximately 1,600 feet (0.3 miles or 0.49 km). It should be noted that the width of the plume may not be accurate because the cell size used in the RT3D model may have also affected the amount of lateral dispersion.

The concentrations of chlorinated solvents measured in the groundwater samples collected from the well at 30 Conestoga Road (8 to 10 mg/L, 1,600 feet from the site, and 165 feet deep) confirm that significant migration of the groundwater plume from the Bishop Tube site has occurred in the fractured bedrock aquifer.

Based upon the calculations performed to determine the cumulative mole fractions of the VOCs contained in the groundwater (see Section 3.7), the chlorinated solvent plume is most likely migrating through the fractured bedrock aquifer by the process of advection (i.e., dissolved solute portion) and the movement of residual separate-phase DNAPLs. Importantly, additional information is required regarding the lateral extent of the dissolved plume and the residual DNAPL body in the subsurface to further refine the RT3D model.

6.0 ASSESSMENT OF POTENTIAL VAPOR INTRUSION INTO ONSITE AND OFFSITE BUILDINGS

On January 24, 2004, the PADEP published their final guidance on "Vapor Intrusion into Buildings from Groundwater and Soil under the Act 2 Statewide Health Standard". This guidance was developed to provide additional screening requirements to assess the potential risks associated with vapor intrusion into buildings originating from VOCs contained in soils and/or groundwater. The new guidance was based upon the recognition that when releases occur near buildings, volatilization of organic compounds in the subsurface can result in the intrusion of VOCs into indoor air.

Per the considerations outlined in the PADEP vapor intrusion guidance document, the following two conditions must be met for the vapor intrusion pathway to be of potential concern: 1) inhabited buildings must be close to a volatile/semivolatile source; and 2) the source concentration must be greater than the threshold or screening concentration. To assess the potential risks to indoor air quality associated with the volatilization of VOCs in soils and groundwater, the new guidance document provides two separate decision matrices (one for soil and one for groundwater), as well as screening tables for soil, groundwater, soil gas, and indoor air quality concentrations.

A horizontal distance of 100 feet between the source (soils or groundwater containing VOCs) and the receptor (i.e., inhabited building) is stipulated as the criterion to define when vapor intrusion should be addressed. In general, a vertical separation distance of 5 feet or more between the source (either soil or groundwater) and the receptor (i.e., inhabited building) must be met to demonstrate that the vapor

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intrusion pathway is incomplete. If separate-phase liquids are present in either the soils or groundwater at a site, the new guidance requires that additional sampling be performed to determine the concentrations of VOCs contained in soil gas or indoor air.

The soil samples collected at the Bishop Tube site during previous investigations (Baker, 2002a and 2003) have determined that elevated concentrations of TCE and other chlorinated VOCs exceeding the PADEP Soil to Groundwater Pathway Standards occur in the subsurface materials (see Figure 12). Locally, soil materials underlying the former vapor degreaser area in Building #8 and the former drum storage area are believed to contain residual separate-phase DNAPLs (Baker, 2003).

The analytical results for the groundwater samples collected during the Supplemental Groundwater Investigation show that the concentration of TCE exceeds the PADEP Non-Residential Statewide Health-based Groundwater Standard in each monitoring well at the Bishop Tube site with the exception of MW01 (upgradient well along the south side of the property) and MW24 (upgradient well along the west side of the property). It should be pointed out that separate-phase DNAPLs are believed to occur in the fractured bedrock aquifer underlying the site in the vicinity of monitoring wells MW25A, MW25B, MW25C, MW26B, MW26C, MW27B, MW28A, MW28B, MW28C, and the well at 30 Conestoga Road (see discussion in Section 3.7).

The Central and Western Chester County Development Authority (CWCCDA) currently owns the Bishop Tube property. The CWCCDA plans to redevelop the Bishop Tube site in the future. The redevelopment plan for the Bishop Tube property is currently undetermined, but may include the re-establishment of industrial operations, the construction of residential housing units, or other commercial uses. Importantly, due to the presence of elevated concentrations of VOCs and probable DNAPLs contained in the soils and groundwater at the Bishop Tube site, the future development plans for the property should address the issue of vapor intrusion into any inhabited buildings (either commercial or residential).

Based upon the horizontal (100 feet) and vertical (5 feet) isolation distances listed in the PADEP vapor intrusion guidance document, the areas where the screening of soil vapor concentrations, indoor air quality monitoring, and/or the installation of mitigation systems should be completed for any future redevelopment at the Bishop Tube site is shown on Figure 56. This map was prepared based upon the concentrations of VOCs detected in the soils and groundwater at the site, assuming a worst case scenario that development would proceed prior to any remediation efforts. To date no soil vapor samples or indoor air quality samples have been collected at the site. It should noted that the PADEP collected soil vapor samples from selected areas at the Bishop Tube site during the summer of 2001. The soil vapor samples were analyzed by the PADEP Bureau of Laboratories onsite mobile laboratory. These soil vapor samples, however, were collected for the purpose of selecting drilling and sampling locations for soil borings, rather than for assessing potential indoor air quality concerns at the site. The boundary line shown on Figure 56 indicates that the development of the southern half of the property would require screening of soil vapor concentrations, collection of indoor air quality samples, and/or the installation of mitigation systems if developed for residential housing purposes. It should be noted that under the procedures of the PADEP vapor intrusion guidance document, the OSHA-derived Permissible Exposure Levels (PELs) may be used upon meeting the notification and monitoring requirements for a "workplace building", in lieu of sampling soil vapor concentrations, collecting indoor air quality samples, and/or installing mitigation systems as required for residential homes. These additional sampling, mitigation, and reporting requirements may be an important consideration regarding the future redevelopment plans for the Bishop Tube property.

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As discussed in Sections 3.5 and 3.7, elevated concentrations of chlorinated solvents and potential DNAPLs occur in the groundwater in the vicinity of monitoring well MW26 (situated in the northeast corner of the Bishop Tube site). An assessment of the of the TCE concentrations measured in the groundwater samples collected from monitoring well MW26 show that the chlorinated solvent plume is continuing to migrate within the fractured bedrock aquifer. The local geologic structure is believed to strongly control the direction of groundwater flow in the bedrock aquifer underlying the Bishop Tube site. This information collectively suggests that chlorinated solvents (namely TCE) may also exhibit an eastward component of migration within the fractured bedrock aquifer. The lateral extent of the dissolved-phase and separate-phase (i.e., DNAPL) fractions of the groundwater plume toward the east has not been determined. It is probable that elevated concentrations of chlorinated solvents along with separate-phase DNAPLs may exist in the fractured bedrock aquifer underlying a portion of the neighboring General Warren Village residential development and commercial properties situated immediately east and northeast of the Bishop Tube site, respectively. Importantly, the concentrations of chlorinated solvents contained in the groundwater may pose an indoor air quality concern to some of the homes in the neighboring General Warren Village residential development and to the local businesses. Additional information regarding the concentrations of chlorinated solvents contained in the shallow groundwater and the depth of groundwater underlying the area east and northeast of the Bishop Tube site is needed to determine whether or not the vapor intrusion pathway into these homes and businesses is incomplete.

7.0   CONCLUSIONS

Based upon the review of available information and the results of the Supplemental Groundwater Characterization, the following conclusions have been developed.

- The geologic horizons beneath the Bishop Tube site can be segregated into three categories: 1) a shallow soil/overburden interval; 2) a weathered bedrock interval; and 3) a deeper unweathered bedrock interval.

- Based upon observations made during the drilling of the monitoring wells and the information obtained from the geophysical well logs, the rock materials underlying the Bishop Tube property were found to consist mainly of phyllite and schist with minor widely separated impure carbonate layers and quartz veins. North of the site the rock materials were found to contain a larger proportion of carbonate material. This facies change of increasing calcareous composition is consistent with published information and personal communications regarding the lithology of the Conestoga formation within the study area.

- Groundwater within the bedrock is produced wholly from fractures. These fractures were found to range from the top of bedrock to a depth of 280 feet (i.e., deepest monitoring well at the site). In general, the occurrence of water-bearing fractures and potential groundwater yields were found to decrease with respect to depth below the ground surface. It should be emphasized that conductive water-bearing fractures are believed to occur in the fractured bedrock aquifer below a depth of 280 feet. Where present, these deep water-bearing fractures may also be functioning as conduits for the migration of chlorinated solvents within the fractured bedrock aquifer.

- Groundwater was found to occur under unconfined and semi-confined conditions. Shallow groundwater within the overburden materials exhibits characteristics of a water table flow system. The groundwater contained in the bedrock was found to occur under semi-confined hydraulic conditions. Based upon the information provided by previous studies performed at the site (Baker 2002b), the weathered bedrock materials are believed to be functioning as a semi-confining layer for the groundwater contained in the bedrock.

- Groundwater flow within the water table and bedrock flow systems during the period between March/April 2003 and February 2004 was toward the north-northeast. This finding is consistent with published information regarding groundwater flow in the area and previous studies performed at the site. The hydraulic gradient for groundwater flow perpendicular to the strike of the foliations contained in the bedrock was found to be primarily horizontal within the central portion of the Bishop Tube property. A strong downward hydraulic gradient was found for groundwater flow parallel to the strike of the rock bedding. This downward flow component is believed be functioning as a supplemental pressure head for the migration of dissolved-phase and separate-phase solutes to deeper levels in the underlying fractured bedrock aquifer.

- Based upon a comparison of water table elevations to surface water elevations, shallow groundwater appears to be serving as a source of baseflow to Little Valley Creek along the eastern portion of the Bishop Tube property. It should be noted that a surface water study completed by the PADEP in May 2003 found elevated concentrations of TCE (i.e., 55 μg/l) in the surface water of Little Valley Creek near the northeast corner of the Bishop Tube property. Groundwater samples collected from monitoring well MW08 (situated near the northeast corner of the Bishop Tube property) have been found to contain elevated concentrations of TCE ranging from 260 μg/l in March/April 2003 to 410 μg/l in October 2003. This information is consistent with the finding outlined above that shallow groundwater is serving as source of baseflow for Little Valley Creek along the eastern portion of the Bishop Tube property.

- FLUTe® NAPL liners were deployed in the boreholes of monitoring wells MW25 and MW27 during the drilling phase of the investigation to determine the presence or absence of separate-phase DNAPLs in the overburden and fractured bedrock materials. The inspection of the FLUTe® NAPL liners deployed in the overburden sections of monitoring wells MW25 and MW27 did not exhibit any signs of separate-phase DNAPLs in the soils and weathered bedrock materials. The FLUTe® NAPL liner deployed in the open bedrock section of monitoring well MW27 also did not display any evidence of separate-phase DNAPLs. A trace amount of staining was observed on the FLUTe® NAPL liner deployed in the open bedrock section of monitoring well MW25, indicating that DNAPLs have invaded the fractured bedrock aquifer at this location to a depth of 280 feet.

- To evaluate changes in the concentrations of chlorinated solvents with respect to depth within the underlying fractured bedrock aquifer, discrete interval groundwater samples were collected during packer tests performed in the boreholes of monitoring wells MW25, MW26, MW27, and MW28. The analytical results for the groundwater samples collected during the packer testing procedures show that the concentrations of chlorinated solvents increase with

respect to depth in the aquifer. This finding is consistent with the supposition that separate-phase DNAPLs may be present in the deeper portions of the bedrock underlying the Bishop Tube property.

- Groundwater samples were collected in March 2003 and October 2003 from the private well at 54 Conestoga Road to evaluate the water quality conditions in the in the aquifer, and to assess the effectiveness of the water treatment system in reducing the concentrations of chlorinated solvents contained in the groundwater. The analytical results show that the concentration of TCE ranged from 19 μg/l to 24 μg/l, exceeding the PADEP Residential Statewide Health-based Groundwater Standard of 5 μg/l. No VOCs were detected in the mid- or post-treatment samples. Accordingly, the carbon treatment system at 54 Conestoga Road appears to be properly lowering the VOC concentrations dissolved in the raw groundwater being supplied by the private well.

- The analytical results for the groundwater samples collected from the monitoring wells at the site during the period between March/April 2003 and February 2004 were found to contain elevated concentrations of 1,1-DCA, 1,2-DCA, cis-1,2-DCE, 1,2-DCE (total), PCE, TCE, and vinyl chloride exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standards. The most portentous and pervasive chlorinated solvent detected in the groundwater samples collected from the monitoring wells is TCE. The analytical results show that the concentrations of TCE exceeded the PADEP Non-Residential Statewide Health-based Groundwater Standard of 5 μg/l in all of the monitoring wells with the exception of monitoring wells MW01 (upgradient well along the south side of the site) and MW24 (upgradient well along the west side of the site). The highest concentrations of TCE were measured in the groundwater samples collected from monitoring wells MW02, MW12, MW13, MW15, MW20, MW22, MW25A, MW25B, MW25C, MW26A, MW26B, MW26C, MW27B, MW28A, MW28B, and MW28C, and the well at 30 Conestoga Road. Based upon the findings of previous investigations performed at the Bishop Tube site (Baker, 2002a, 2002b, 2003), the source of the chlorinated solvents contained in the groundwater is attributed to releases of DNAPLs from the former vapor degreaser area in Building #5, the former vapor degreaser area in Building #8, and the former drum storage area.

- Elevated concentrations of MTBE exceeding the PADEP Non-Residential Statewide Health-based Groundwater Standard were detected in the groundwater samples collected from selected monitoring wells at the Bishop Tube site. MTBE is an anti-knock and oxygenate supplement that is added to unleaded gasoline. Based upon historical information, this organic solvent was not used in the industrial processes performed at the Bishop Tube site. The source of the MTBE in the groundwater underlying the Bishop Tube property is believed to be a release of hydrocarbon compounds at the Mobil bulk oil terminal situated immediately west of the site. Based upon its high solubility, MTBE typically migrates at the same rate as groundwater flow, behaving as a conservative tracer. The presence of MTBE in the groundwater at the Bishop Tube site indicates that the gasoline additive has migrated laterally along the strike of the rock beds. This finding suggests that other dissolved solutes (i.e., chlorinated solvents) may also be migrating laterally along the strike of the rock beds in the underlying fractured bedrock aquifer. This supposition is important because it suggests that dissolved-phase and separate-phase solutes originating from the Bishop Tube site may have

migrated laterally along strike within the bedrock aquifer toward the commercial and residential properties situated east and northeast of the site.

- Low concentrations of 1,4-dioxane were detected in monitoring wells MW25A (13 µg/l), MW28C (6 µg/l), and the well at 30 Conestoga Road (8 µg/l). The concentrations of 1,4-dioxane measured in the groundwater samples collected from these wells were below the PADEP Non-Residential Statewide Health-based Groundwater Standard established for this compound. 1,4-dioxane is a stabilizer, that is added to chlorinated solvents (namely TCE and 1,1,1-TCA) as a corrosion inhibitor. Like MTBE, 1,4-dioxane is very mobile in the subsurface environment, and tends to migrate at the front of chlorinated solvent groundwater plumes. The presence of 1,4-dioxane in the groundwater samples collected from the monitoring wells suggests that solvents containing this stabilizer were used in the past at the Bishop Tube facility. The relatively low concentrations of 1,4-dioxane detected in the groundwater samples suggest that the leading edge of the chlorinated solvent plume originating from the Bishop Tube site has very likely migrated past the location of the well at 30 Conestoga Road.

- The analytical results show that the following heavy metals were detected in the groundwater samples collected from the monitoring wells at "dissolved" concentrations that exceeded the PADEP Non-Residential Statewide Health-based Groundwater Standards: chromium, lead, nickel, and thallium. In general, the highest "dissolved" concentrations of heavy metals were detected in the groundwater samples collected from the monitoring wells situated along the eastern portion of the Bishop Tube site in the vicinity of the former drum storage area (i.e., monitoring wells MW22 and MW27A), the former cesspool area (i.e., east of the incinerator room in Building #5 near monitoring well MW21), and the former waste lagoon (i.e., southeastern corner of Building #8 near monitoring wells MW04, MW05, MW07, and MW09). Based upon historical records of the former plant operating procedures, heavy metal liquid wastes and sludges were processed, stored, and/or disposed of in each of these areas. This information suggests that the source of the heavy metals contained in the groundwater along the eastern portion of the Bishop Tube site is most likely related to the former industrial and waste handling procedures performed at the facility.

- Elevated concentrations of fluoride that exceeded the PADEP Non-Residential Statewide Health-based Groundwater Standard were consistently detected in the groundwater samples collected from monitoring wells MW04, MW05, MW07, MW09, MW21, MW22, MW23, MW25A, MW26A, MW26B, MW27A, and MW27B. Elevated concentrations of nitrate that exceeded the PADEP Non-Residential Statewide Health-based Groundwater Standard were measured in the groundwater sample collected from monitoring well MW07 during the February 2004 sampling event. In general, the monitoring wells containing elevated concentrations of fluoride and nitrate are situated along the eastern portion of the Bishop Tube site in the vicinity of the former drum storage area (i.e., monitoring wells MW22, MW27A, and MW27B), the former cesspool area (i.e., east of the incinerator room in Building #5 near monitoring well MW21 and MW23), and the former waste lagoon (i.e., southeastern corner of Building #8 near monitoring wells MW04, MW05, MW07, MW09, MW26A, MW26B). Based upon historical records of the former plant operating procedures, hydrofluoric and nitric acid pickle liquor was processed, stored, and/or disposed of in each of

these areas. This information suggests that the source of the fluoride and nitrate contained in the groundwater along the eastern portion of the Bishop Tube site is most likely related to the former industrial and waste handling procedures performed at the facility.

- During each quarterly groundwater sampling event an electronic interface probe was used to check the monitoring wells at the Bishop Tube site for the presence of separate-phase LNAPLs and DNAPLs. In addition, discrete top and bottom samples were collected from selected monitoring wells to determine the presence/absence of separate-phase liquids. No separate-phase LNAPLs were observed in the monitoring wells during the time period between March 2003 and February 2004. During the February 2004 sampling event, separate-phase DNAPLs were observed in monitoring wells MW25A, MW25B, MW26A, MW26B, MW27B, MW28A, MW28B, and MW28C. Based upon the principals of effective solubility (i.e., calculated cumulative mole fractions), separate-phase DNAPLs are suspected to be present in the vicinity of monitoring wells MW02, MW25A, MW25B, MW25C, MW26A, MW26C, MW27B, MW28A, MW28B, MW28C, and the well at 30 Conestoga Road. The sampling data suggest that separate-phase DNAPLs have invaded the fractured bedrock aquifer underlying the Bishop Tube property to depths ranging from 24 to 280 feet below the ground surface. The total depth of DNAPL invasion in the fractured bedrock aquifer underlying the site remains undetermined. Based upon the concentrations of VOCs measured in the groundwater samples and the calculated cumulative mole fractions, the separate-phase DNAPL portion of the groundwater plume is believed to be approaching the well at 30 Conestoga Road (i.e., located approximately 1,600 feet northeast of the Bishop Tube site). Due to the paucity of offsite monitoring wells, the lateral extent of the separate-phase DNAPL portion of the groundwater plume within the fractured bedrock aquifer remains undetermined.

- The groundwater samples collected during the February 2004 sampling event were analyzed for radon to estimate NAPL saturations in the bedrock aquifer underlying the Bishop Tube site. The calculations show that the potential DNAPL saturations in the bedrock materials surrounding monitoring wells MW02, MW25A, MW25B, MW25C, and MW28C range from 0.03% to 40.79%. This later value was calculated using the radon concentrations in the groundwater sample collected from monitoring well MW28C. Although separate-phase DNAPLs have been observed in monitoring well MW28C, a saturation value of 40.79% may be an overestimate of the actual conditions within the fractures of the bedrock surrounding this well. The calculated NAPL saturation values were used to determine the potential volume of residual DNAPLs contained in the fractured bedrock aquifer surrounding monitoring wells MW02, MW25A, MW25B, MW25C, and MW28C. The calculations show that the volume of residual DNAPLs contained in the bedrock surrounding monitoring well MW02 may range from 51 to 103 gallons. Calculations performed for monitoring wells MW25, MW26C, and MW28C show that the following volumes of residual DNAPLs may be present within the fractures of the bedrock surrounding these wells: 4,800 to 9,600 gallons (MW25), 290 to 580 gallons (MW26C), and 1,920 to 3,835 gallons (MW28C). If should be pointed out that problems were encountered when shipping the groundwater samples collected for the analysis of radon to testing laboratory. Accordingly, the NAPL saturations determined by this evaluation are considered only rough estimates.

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- To evaluate the migration of the chlorinated solvent groundwater plume in the fractured bedrock aquifer underlying the Bishop Tube site, a comparison was made between: 1) changes in the concentration of TCE with respect to fluctuations of groundwater levels; and 2) changes in the concentration of TCE over time. This assessment shows that a stable to decreasing trend in the TCE concentrations of the groundwater plume is occurring for monitoring wells MW25A, MW25C, and MW26A. The stable to decreasing trend for the TCE concentrations measured in the groundwater samples collected from monitoring wells MW25A, MW25C, and MW26A suggests that the center of mass for the groundwater plume may have already migrated past the screened intervals of these wells. Conversely, the concentrations of TCE were found to be increasing in monitoring wells MW22, MW26C, and MW28C. The positive slopes indicated by the trend analyses for monitoring wells MW26C and MW28C suggests that the dissolved-phase and separate-phase portions of the TCE groundwater plume is continuing to migrate within the deeper portion of the fractured bedrock aquifer in the vicinity of these screened intervals. In addition, the trend analyses for monitoring wells MW28A and the well located at 30 Conestoga Road suggest that the groundwater plume containing TCE is continuing to migrate in a north-northeasterly direction. These findings are consistent with the results of the RT3D groundwater flow model showing that the dissolved-phase and separate-phase chlorinated solvent plume is migrating in a north-northeast direction from the Bishop Tube site.

- A three-dimensional groundwater flow model using MODFLOW 2000 was developed to more fully evaluate groundwater flow within the fractured bedrock aquifer in the vicinity of the Bishop Tube site. A MODPATH pathline analysis was also run using the calibrated steady-state MODFLOW solution. The simulated pathlines generated by MODPATH show that particles in model layers 1, 2, and 3 (i.e., 0 to 300 feet below the ground surface) have reached and are discharging groundwater to Little Valley Creek between 2,100 and 5,500 feet northeast of the Bishop Tube site. Groundwater contained in the deeper portion of the bedrock aquifer (i.e., model layer 4, situated between 300 and 400 feet below the ground surface) has traveled an estimated 3,200 feet northeast of the Bishop Tube site, and will ultimately discharge groundwater to Little Valley Creek in the future. To evaluate the groundwater modeling results, the areas where the simulated pathlines are providing baseflow to Little Valley Creek were compared to the analytical results for the surface water samples collected from the stream northeast of the Bishop Tube site. This comparison demonstrated a close match between the locations where elevated concentrations of TCE were detected in the surface water samples and areas where the simulated pathlines of the groundwater flow model are providing baseflow to Little Valley Creek. Importantly, these findings suggest that impacts to the water quality of Little Valley Creek (an "*Exceptional Value*" stream) and the indigenous fauna and flora may be greatest immediately adjacent to the Bishop Tube site and along the more distant downstream section of the stream where deep groundwater originating from the Bishop Tube facility is discharging via baseflow.

- The results of the RT3D groundwater model simulation show that TCE and its daughter products cis-1,2-DCE and vinyl chloride are migrating within the bedrock aquifer from the Bishop Tube site toward the northeast. Based upon the model simulations, TCE, cis-1,2-DCE, and vinyl chloride are estimated to have migrated up to 3,000 feet, 3,200, and 3,500 feet, respectively, in the bedrock aquifer at concentrations exceeding the PADEP Non-Residential

Statewide Health-based Groundwater Standards. Based upon the limited number of offsite wells, the lateral extent (i.e., toward the east along strike) and downgradient extent (i.e., toward the northeast parallel to the regional groundwater flow direction) of the chlorinated solvent groundwater plume originating from the Bishop Tube site remains undetermined.

- An assessment was made comparing the soil and groundwater analytical results for the Bishop Tube site to the recommended offset distances outlined in the final PADEP guidance on "Vapor Intrusion into Buildings from Groundwater and Soil under the Act 2 Statewide Health Standard". This comparison showed that the development of the southern half of the property would require screening of soil vapor concentrations, collection of indoor air quality samples, and/or the installation of mitigation systems if developed for residential housing purposes.

- Based upon the hydraulic characteristics of the underlying fractured bedrock aquifer, a portion of the chlorinated solvent groundwater plume originating from the Bishop Tube site is suspected to be migrating in an eastward direction. The lateral extent of the dissolved-phase and separate-phase (i.e., DNAPL) fractions of the groundwater plume toward the east has not been determined. It is probable that elevated concentrations of chlorinated solvents along with separate-phase DNAPLs may exist in the fractured bedrock aquifer underlying a portion of the neighboring General Warren Village residential development and commercial properties situated immediately east and northeast of the Bishop Tube site, respectively. Importantly, the concentrations of chlorinated solvents contained in the groundwater may pose an indoor air quality concern to some of the homes in the neighboring General Warren Village residential development and to the local businesses. Additional information regarding the concentrations of chlorinated solvents contained in the shallow groundwater and the depth of groundwater underlying the area east and northeast of the Bishop Tube site is needed to determine whether or not the vapor intrusion pathway into these homes and businesses is incomplete.

8.0   RECOMMENDATIONS

The primary objectives of the Supplemental Groundwater Investigation performed at the Bishop Tube site were to: 1) evaluate the characteristics of the fractured bedrock aquifer underlying the site; 2) determine the presence or absence of separate-phase DNAPLs occurring within the fractured bedrock aquifer; 3) evaluate changes in the concentrations of chlorinated solvents with respect to depth within the fractured bedrock aquifer; 4) assess seasonal fluctuations in the elevation of groundwater and the concentrations of chlorinated solvents dissolved in the fractured bedrock aquifer; and 5) evaluate the characteristics of groundwater flow and the migration of chlorinated solvents dissolved in the fractured bedrock aquifer by developing a groundwater flow model. The recommendations outlined below stem from the conclusions presented in Section 7.0.

The findings of the Supplemental Groundwater Investigation show that the groundwater underlying the Bishop Tube property and the area situated northeast of the site contains elevated concentrations of TCE and TCE breakdown products. Based upon the information collected during the investigation, separate-phase DNAPLs are suspected to be present within the bedrock aquifer underlying the Bishop Tube property and the area situated east-northeast of the site. The dissolved-phase and separate-phase fractions of the groundwater plume are continuing to migrate within in the bedrock aquifer in a northeast direction.

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Based upon the limited number of offsite wells, the lateral extent (i.e., toward the east along strike) and downgradient extent (i.e., toward the northeast parallel to the regional groundwater flow direction) of the chlorinated solvent plume remains undetermined. Groundwater elevation measurements show that shallow groundwater along the eastern portion of the Bishop Tube site is discharging as baseflow to Little Valley Creek. The results of the groundwater flow model suggest that the groundwater contained in the deeper portion of the bedrock aquifer is discharging as baseflow to Little Valley Creek farther northeast of the Bishop Tube site. Based upon the findings of previous investigations performed at the Bishop Tube site (Baker, 2002a, and 2003), the source of the chlorinated solvents contained in the groundwater is attributed to releases of DNAPLs from the former vapor degreaser area in Building #5, the former vapor degreaser area in Building #8, and the former drum storage area. It is expected that the residual DNAPLs contained in the soils and weathered bedrock materials underlying these source areas will continue to serve as a supplemental source for the dissolved-phase chlorinated solvents (namely TCE) within the bedrock aquifer. To mitigate these residual sources, Baker (2003) had previously recommended that the Department consider implementing a Feasibility Study to evaluate different remedial techniques to reduce the concentrations of chlorinated solvents contained in the soils.

1. The untreated groundwater samples collected from the private well at 54 Conestoga Road have been found to contain elevated concentrations of TCE exceeding the PADEP Residential Statewide Health-based Groundwater Standards. The results of the groundwater flow model suggest that the TCE dissolved in the groundwater of the private well at 54 Conestoga Road is originating from the Bishop Tube site. The Department has been collecting water samples from this well on a semi-annual basis to evaluate the effectiveness of the carbon filtration system for reducing the concentrations of chlorinated solvents dissolved in the raw groundwater. Baker recommends that the Department continue to collect pre-, mid-, and post-treatment water samples from the private well at 54 Conestoga Road on a semi-annual basis until: 1) an alternative source of potable water is furnished to the persons living in this home, or 2) a remedial program is implemented that reduces the dissolved concentrations of chlorinated solvents below the PADEP Residential Statewide Health-based Groundwater Standards. The semi-annual groundwater samples should be analyzed for the following selected target chlorinated VOCs: 1,1-DCA, 1,1-DCE, cis-1,2-DCE, trans-1,2-DCE, PCE, 1,1,1-TCA, TCE, and vinyl chloride.

2. Based upon the findings of the Supplemental Groundwater Investigation, Baker recommends that the Department perform an ecological assessment of the existing conditions in Little Valley Creek both adjacent to and downgradient of the Bishop Tube site. The information provided by the ecological assessment will be used to evaluate impacts of the chlorinated solvents and other inorganic constituents of environmental concern (i.e., fluoride, nitrate, and heavy metals) on the indigenous flora and fauna in Little Valley Creek (classified as an "*Exceptional Value*" stream).

3. Based upon a comparison of water table elevations to surface water elevations, shallow groundwater appears to be serving as a source of baseflow to Little Valley Creek along the eastern portion of the Bishop Tube property. Based upon this finding, Baker recommends that the Department consider performing an investigation to define the lateral limits of where the shallow groundwater plume containing chlorinated solvents is discharging (via baseflow) to Little Valley Creek along the eastern edge of the Bishop Tube property. This information will be used to evaluate remedial options for mitigating the discharge of chlorinated solvents to this "*Exceptional Value*" stream. The pathlines generated by the groundwater flow model show that groundwater contained in the deeper portion of the bedrock aquifer is being discharged via baseflow to Little

Valley Creek 2,100 to 5,500 feet northeast of the Bishop Tube site. The analytical results for the surface water samples collected by the PADEP in the spring of 2003 indicate that the concentrations of TCE increase in Little Valley Creek within the predicted groundwater discharge area. Based upon this finding, remediation of the DNAPLs contained in the deeper portion of the fractured bedrock aquifer underlying the Bishop Tube site may be necessary to reduce the discharge of chlorinated solvents to Little Valley Creek via baseflow from the deep groundwater flow system.

4. Based upon the limited number of offsite wells, the lateral extent (i.e., toward the east along strike) and downgradient extent (i.e., toward the northeast parallel to the regional groundwater flow direction) of the chlorinated solvent groundwater plume remains undetermined. It is probable that elevated concentrations of chlorinated solvents are contained in the shallow and deep portions of the bedrock aquifer underlying the General Warren Village residential development and commercial properties situated east and northeast of the Bishop Tube site, respectively. Separate-phase DNAPLs are also suspected to be present in the deep portion of the bedrock aquifer in the areas east and northeast of the site. To evaluate the potential risks to the neighboring residents and businesses, Baker recommends that the Department consider drilling an additional offsite monitoring well (MW29). Monitoring well MW29 should be drilled northeast of the site to evaluate the concentrations of chlorinated solvents contained in the shallow and deep portions of the fractured bedrock aquifer underlying the commercial business along Business Route 30. The suggested drilling location for this additional offsite monitoring well is shown on Figure 57. Monitoring well MW29 should be drilled to a depth of 250 feet and designed using three separate wellscreen intervals (i.e., multi-screened well). This design will provide information regarding changes in the concentrations of chlorinated solvents with respect to depth within the aquifer. In addition, monitoring well MW29 will provide data regarding the lateral (i.e., along strike) and northeast migration of the dissolved-phase and separate-phase fractions of the chlorinated solvents contained in the bedrock aquifer.

    As mentioned above, the downgradient extent of the groundwater plume also remains undetermined. Baker recommends that the Department also consider drilling an additional deep offsite well (MW30) farther northeast of the site to evaluate the downgradient extent of the chlorinated solvent groundwater plume in the fractured bedrock aquifer. The suggested drilling location for this additional offsite well is shown on Figure 57. Monitoring well MW30 should be drilled to a depth of 250 feet and designed using three separate wellscreen intervals (i.e., multi-screened well). This design will provide information regarding changes in the concentrations of chlorinated solvents with respect to depth within the aquifer. The testing results for the groundwater samples collected from this additional offsite well will be used to evaluate the water quality conditions in the aquifer, to refine the groundwater flow model, and to assess potential risks to downgradient receptors.

5. The results of the groundwater flow model suggest that both dissolved-phase and separate-phase chlorinated solvents are migrating within the fractured bedrock aquifer in a northeast direction from the Bishop Tube site. The migration of the chlorinated solvent groundwater plume has the potential to impact receptors (i.e., private wells and Little Valley Creek) situated hydraulically downgradient of the study area. Based upon this information, Baker recommends that the Department consider implementing a Feasibility Study to evaluate different remedial techniques for reducing the concentrations of chlorinated solvents contained in the fractured bedrock aquifer.

This Feasibility Study should address the following issues: 1) the high concentrations of chlorinated solvents contained in the groundwater that may inhibit the use of bioremedial techniques; 2) the removal/abatement of separate-phase DNAPLs within the bedrock aquifer; 3) access problems in regards to delivering chemical oxidants and/or amendments to the subsurface (especially for the areas inside Buildings #5 and #8 as well as offsite areas); 4) the compatibility of the remediation program with the future development plans for the site; and 5) a comparison of costs between different remedial techniques. A discussion of potential groundwater remedial technologies is provided in Appendix H.

6. The subsurface conditions at the Bishop Tube site are not ideal for remediation because the chlorinated solvent groundwater plume occurs within a fractured bedrock aquifer. The fractured bedrock aquifer is complex geologically and exhibits large fluctuations in permeability (i.e., hydraulic conductivity). These fluctuations in hydraulic conductivity reflect groundwater flow through the heterogeneous fracture network contained in the rocks comprising the aquifer. Due to these site conditions, Baker recommends that the Department consider performing a cross-hole geophysical survey (i.e., resistivity, seismic, and/or radar) between selected monitoring wells for collecting information regarding the location, configuration, extent, and connectivity of the fractures contained in the bedrock. This information will ultimately be used to select applicable remediation technologies and to better define target zones for focusing remediation efforts.

7. During the interim period when a decision regarding the cleanup objectives for the groundwater plume is being evaluated, Baker recommends that the Department consider collecting groundwater samples from selected onsite and offsite monitoring wells on a semi-annual basis. This information will be used to evaluate and monitor the migration of the chlorinated solvent plume in the aquifer. Static water levels in the onsite and offsite monitoring network as well as possibly other non-pumping wells in the area (i.e., industrial complex wells) should be performed on a quarterly basis. The thickness of separate-phase DNAPLs should also be measured on a quarterly basis.

    To reduce project costs, Baker recommends that the following twenty-two monitoring wells be included in the semi-annual groundwater monitoring program: MW01, MW02, MW04, MW07, MW09, MW12, MW15, MW21, MW22, MW23, MW25A, MW25B, MW25C, MW26A, MW26B, MW26C, MW27A, MW27B, MW28A, MW28B, MW28C, and the well at 30 Conestoga Road. These monitoring wells are recommended for the semi-annual monitoring program based upon the elevated concentrations of chlorinated solvents and inorganic compounds (i.e., heavy metals, fluoride, and nitrate) contained in the groundwater. To evaluate the concentrations of chlorinated solvents and other important biological indices, Baker recommends that the groundwater samples collected from the monitoring wells during the semi-annual groundwater sampling program be analyzed for the following parameters: selected target chlorinated VOCs (1,1-DCA, 1,1-DCE, cis-1,2-DCE, trans-1,2-DCE, MTBE, PCE, 1,1,1-TCA, TCE, and vinyl chloride); dissolved gases (methane, ethane, and ethene); selected total and dissolved metals (i.e., chromium, lead, nickel, and thallium); chloride, sulfate, sulfite, fluoride, nitrate, total organic carbon (TOC). In addition, Baker recommends that the following parameters be measured in the monitoring wells at the time the groundwater samples are collected:

8. As more groundwater quality and elevation data are gathered, Baker recommends that the Department consider updating/modifying the three-dimensional groundwater flow model

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developed for the project area. The additional data will be used to refine the groundwater flow model, that may ultimately be used for evaluating different remedial techniques for the chlorinated solvents contained in the groundwater plume (if deemed necessary).

If you have any questions, please do not hesitate to contact me at (717)-221-2019, or Mr. Raymond Wattras, our GTAC Program Manager, at (412) 269-2016.

Sincerely,

BAKER ENVIRONMENTAL, INC.

Mark B. Ioos, P.G.
Baker Project Manager/Senior Geologist

MBI:tme
Attachments
cc: Mr. Tim Sheehan – PADEP HSCA Supervisor
         Mr. Doug Cordelli – PADEP GTAC 3 Contract Manager

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