

Section 3

Environmental Conditions at the Bannister Federal Complex

3.0 Introduction.....	3-1
3.1 Alluvial Hydrostratigraphy and Geology.....	3-1
3.2 Bedrock Stratigraphy and Geology.....	3-4
3.3 Groundwater Flow Systems.....	3-6
3.4 Impacts to Groundwater Flow	3-7
3.4.1 Building Footing Tile Drains	3-7
3.4.2 Cattail Area	3-8
3.5 Velocity of Groundwater at the BFC.....	3-9
3.5.1 BRGFS	3-9
3.5.2 ICGFS	3-9
3.6 BFC Hydrological Conceptual Model	3-10
3.7 BFC Contaminants.....	3-12
3.8 Chlorinated Solvents – Behavior in the BFC Subsurface Environment	3-16
3.8.1 TCE.....	3-16
3.8.1.1 History of TCE/Use at the BFC.....	3-17
3.8.2 Aromatic Compounds	3-20
3.9 Conceptual Site model Groundwater Contamination	3-21
3.9.1 Fit of the model to the KCP.....	3-23
3.10 Potential for Contaminant Migration in Bedrock	3-23
3.11 Application of Natural Attenuation Protocol.....	3-26
3.12 Natural Attenuation at the Groundwater/Surface Water (Indian Creek/Blue River) Interface	3-29
3.12.1 Surface Water/ Groundwater Interaction	3-29
3.12.1.1 Contaminant Concentrations at the Aquifer/Surface Water Interface	3-32
3.12.2 Relevant Field Measurements at the BFC	3-33
3.13 Specific Conceptual Models for the Origin and Movement of Contamination in the BRGFS and ICGFS.....	3-36
3.13.1 BRGFS	3-36
3.13.2 ICGFS	3-38
3.14 Potential Co-solvent Effects	3-39
3.15 Conceptual Model of Contamination.....	3-40
3.16 Groundwater Use	3-41
3.17 Current Surface Water	3-45

3.0 Introduction

This section describes the subsurface material (soil and rock) that exists at the BFC. The physical and chemical properties of this material and how the material properties influence the flow of water (and contamination) through it.

The nature and type of contamination found at the BFC is also discussed summarizing efforts to determine how this contamination behaves in the BFC subsurface environment.

3.1 Alluvial Hydrostratigraphy and Geology

During the Ice Age, glaciers extended southward into what is now northern Kansas City. These glaciers and the changes wrought by their melting are responsible for the relatively flat topography of the area. Characteristic east-facing rock escarpments are held up by rocks comprising the Kansas City Group generally composed of alternating limestones and shales. The gentle slopes and lowlands intervening between the escarpments are largely underlain with shales.

The BFC is built on the alluvial soils of the Blue River flood basin described as dark, clayey and loamy. The alluvium is approximately 40-45 ft thick and includes a continuous upper layer of thin-bedded clayey silt, with minor amounts of sand and a basal gravel within a sand-silt-clay matrix. The basal gravel, ranging in thickness from a few inches to 8 ft, consists of fragments of eroded bedrock in a sand-silt-clay matrix and is likewise continuous throughout the site. These two layers are separated in certain areas by a layer of olive to blue-green clayey silt. All three layers transmit water and constitute the alluvial aquifer. Some secondary permeability features exist due to small burrow and root zones. These features, however, do not extend to significant depths within the aquifer. In many parts of the facility, fill material has been added over the years and now

comprises near surface material. This is especially true in areas in northeast portions of the BFC.

Groundwater monitoring wells at the BFC are screened to sample groundwater in the upper portion of the alluvium to monitor for potential floating contaminants and shallow contaminant sources. The basal gravel in the lower alluvium, at the contact with the underlying bedrock, is also monitored because it is the most conductive water bearing zone. Wells that monitor this zone are called “lower completion” wells while wells that monitor the upper zone are called “upper completion” wells. These wells monitor contaminants that have a specific gravity greater than that of water (e.g., trichloroethylene). An installation diagram of typical wells is provided as Figure 3.1 . It shows the upper and lower completions as well as the composition of the alluvium typical at the site. The distribution of the alluvium at the site is also shown by the fence diagram for the TCE Still Area provided as Figure 3.2 and for areas in the eastern portion of the BFC in the fence diagram provided as Figure 3.3

Bail, slug and pumping tests have been conducted on numerous BFC wells since the inception of characterization in 1984 through 2000 to determine the hydrologic characteristics of the alluvial aquifer. This section provides a summary and discussion of those results. Analysis of bail and slug tests are the same; the difference is in how water within the well is displaced during data collection. For bail tests water is rapidly bailed from the well and the rate at which well water levels recover is measured. For slug tests, a solid slug is introduced into the well causing an instantaneous rise in water levels within the well and the temporal decline in water levels is measured. The test is repeated after complete water level recovery by removing the slug and measuring water level recovery. For both bail and slug tests hydraulic conductivity is a function of the rate at which water levels recover to ambient conditions for a specified well geometry (well diameter, length of screen, etc.).

Bail and slug test results at the BFC show upper clayey-silt hydraulic conductivities typically ranging from 0.1 ft/d to 10 ft/d with both higher and lower values occasionally being measured. Basal gravel results show typical hydraulic conductivity results ranging from 1 ft/d to 100 ft/d (Tables 3.1 to 3.6). Similar to the upper clayey-silt, values outside of the typical basal gravel hydraulic conductivity range have been recorded. It should be noted that there is a dichotomy in testing results. Bail tests conducted early in the BFC characterization effort measured similar upper clayey-silt and basal gravel hydraulic conductivities which suggested similar hydraulic characteristics (Korte et al. 1985, Madril et al. 1986). Subsequent slug testing yielded upper clayey-silt and basal gravel hydraulic conductivities that varied by an order of magnitude with the basal gravel having the higher hydraulic conductivity. The later slug test results are a more accurate representation of the range and distribution of hydraulic conductivity at the BFC.

In addition to bail and slug tests, eight constant rate pumping tests have been conducted at the BFC. Pumping tests involve pumping a well at a constant rate and measuring the rate and magnitude of induced drawdown in the pumping and adjacent observation wells. The observed hydraulic response is compared against type curves to determine hydraulic conductivity and storativity in the vicinity of the pumping well. Initial pumping tests conducted at the BFC utilized pumping wells with screens spanning the entire saturated thickness (both basal gravel and upper clayey silt) and the hydraulic conductivity results (1.1 ft/d to 3.0 ft/d) characterize the bulk hydraulic conductivity of both. Later pumping tests specifically targeted the basal gravel or upper clayey-silt and yielded basal gravel hydraulic conductivity values of between 14 ft/d to several 100's ft/d and upper clayey-silt hydraulic conductivity values of between 1 ft/d to 10's ft/d.

In summary, upper clayey-silt and basal gravel hydraulic conductivities typically range from 0.1 ft/d to 10 ft/d and 1 ft/d to 100 ft/d, respectively.

3.2 Bedrock Stratigraphy and Geology

The BFC is located in the Great Plains physiographic province along the southeastern flank of the Forrest City basin. Strata underlying the site consists of approximately 45 ft of unconsolidated Quaternary alluvium that rests on some 2400 ft of Paleozoic strata. A Precambrian crystalline basement complex underlies the Paleozoic section. In descending order, the rock strata consist of the following:

- Pennsylvanian: approximately 500 ft of green-to-gray shale, siltstone, and sandstone with thin lenses of coal grading downward into offshore shales and carbonates.
- Mississippian: approximately 400 ft of gray limestone, cherty limestone, dark gray shale, dolomite, siltstone, and sandstone.
- Devonian: approximately 200 ft of dark-gray shale, bedded chert, and limestone.
- Ordovician: approximately 700 ft of dark-gray dolomites and sandstones.
- Cambrian: approximately 600 ft of dark-gray dolomites, shales, and sandstones.
- Precambrian: crystalline basement complex.

The Pennsylvanian bedrock (present nearest to land surface at the BFC) is notable for its uniformity (Heckel and Mitchell 1978). Thus, there are no structural features, such as faults, that affect the BFC. Based on a structural contour map of the Gasconade Dolomite for west-central Missouri (Gann et al. 1974), the bedrock dips in a northwest direction at 15 to 20 ft/mile. The bedrock strike is in a northeast-southwest direction.

The bedrock underlying the alluvium at the BFC consists of shales and sandstones of the Pleasanton Group (Figure 3.4). The erosional surface of the Pleasanton Group slopes gently to the east towards the Blue River (Figure 3.5.) The overlying sandstones, shales and limestones of the Kansas City Group have been eroded away and are no longer present at the site.

The Knobtown Sandstone within the Pleasanton Group underlies the alluvium across the central portion of the KCP (Figure 3.6). This sandstone is a well-sorted, very fine-grained, well-cemented, lithic arkose. Generally, the Knobtown consists of monocrystalline quartz, sedimentary rock fragments, authigenic clay, potassium feldspar, plagioclase, chlorite from altered biotite, muscovite, and carbonaceous material. Results of thin-section analyses indicate approximately 12% intergranular pore space (Korte et al. 1985). The Knobtown ranges in thickness from approximately 5 ft to 10 ft and is present in the upper 30 ft of the Pleasanton Group except where it has been removed by Quaternary erosion (DOE 1984). The surrounding unnamed shales of the Pleasanton Group are present in the remaining portions of the site. These shale units are comprised of dense, indurated shale that show no evidence of secondary permeability. The shale's are dark green to gray and occasionally contain light-green sand stringers. Another sandstone unit, the Hepler sandstone, is found as a part of the bedrock at the BFC. Approximately 20 feet of unnamed shale are present over the Hepler sandstone at the BFC with at least 20 feet of additional shale present below the Hepler based on logs derived from bedrock wells at the facility (Kayenta, 2002).

3.2.1 Hydraulic Properties, Bedrock

Packer tests conducted on bedrock wells indicate that the bedrock has very low hydraulic conductivity. For example, packer tests conducted on the Pleasanton Shale showed no-flow conditions. Because packer tests can measure hydraulic conductivity's as low as 0.0001 ft/d (3.5×10^{-8} cm/s), it is safe to assume that the hydraulic conductivity of the Pleasanton shale is 0.00001 ft/d (3.5×10^{-9} cm/s) or lower. Based on packer tests of former bedrock wells 21, 22, and 23 the hydraulic conductivity of the Knobtown Sandstone ranges from 0.005 to 0.04 ft/d (1.8×10^{-6} to 1.4×10^{-5} cm/s) (Korte et al. 1985; Madril et al. 1986). Bail tests of wells 122, 123, and 124, analyzed using Hvorslev's method, produced similar measurements to Knobtown Sandstone hydraulic conductivities (2×10^{-5} to 4×10^{-5} cm/s) (DOE 1990a).

Because the Knobtown is in hydraulic communication with the overlying alluvium, groundwater flow rates are of concern. Using the same hydraulic gradient as the overlying alluvium (0.005 to 0.01 ft/ft) and an assumed effective porosity of 9% (total porosity = 12 %) (Korte et al 1985) groundwater migrates at a rate of 0.16 to 1.3 ft/yr in the Knobtown sandstone.

The Hepler Sandstone exhibited no-flow conditions in well 22 and 0.024 ft/d (8×10^{-6} cm/s) in well 23. This indicates that the Hepler Sandstone is a low-permeable formation. The low hydraulic conductivity of the Hepler Sandstone combined with the hydraulic isolation created by 35 ft of impermeable Pleasanton Shale separating the alluvium and the Hepler Sandstone indicates that contamination of this unit from above is not a concern.

In summary the bedrock beneath the BFC alluvium is impermeable and the bedrock surface can be considered the bottom of the KCP groundwater flow system.

3.3 Groundwater Flow Systems

The BFC groundwater flow system consists of two parts, the Indian Creek Groundwater Flow System (ICGFS) and the Blue River Groundwater Flow System (BRGFS) (Figure 3.7). The BRGFS is defined as that portion of the KCP containing groundwater that discharges to the Blue River. The other flow system is called the ICGFS and under non-industrialized conditions, groundwater within this flow system would discharge solely to Indian Creek. Because of industrialization, some of the groundwater within this flow system discharges to building footing tile drains (see also Section 3.4.1).

3.4 Impacts to Groundwater Flow

3.4.1 Building Footing Tile Drains

The purpose of the building-footing tile drain systems at the BFC are to keep groundwater from infiltrating into building basements. The footing tile drains consist of perforated pipe installed along and slightly below the outside of basement walls. Groundwater seeps into the tile drain and flows by gravity to a sump where it is removed by a pump to either the sanitary sewer system or groundwater treatment system. The net result is that the water table in the vicinity of a basement is maintained at a level below the basement floor. Figure 3.8 shows the location and depths of the drains in the MMB. The figure also shows the flow rate measured from the sump and whether the groundwater discharging to the sump contained VOCs.

Figures 3.9 and 3.10 show clayey-silt (upper completion) and basal gravel (lower completion) historic average potentiometric surfaces. Figures 3.11 and 3.12 show groundwater elevations based on sampling conducted in the fall of 2012. The surfaces show that the basement footing tile drains continue to depress both the upper clayey silt and basal gravel potentiometric surfaces. The surfaces also show elevated water levels associated with leaking underground fire protection and/or water supply lines in the industrialized parts of the BFC. This is further discussed in Annual Groundwater Corrective Action Reports (DOE, 2012). The existence of footing tile drains serves to limit the flow of groundwater southward toward Indian Creek.

DOE conducted a study in 2010 to determine the effects on the flow of groundwater under various future potential building teardown scenarios (Laase 2010b). To mimic the various decommissioning scenarios, model input parameters and boundary conditions were systematically altered which included removing leakage from underground utilities, partially and/or completely removing building footing tile drains and removing pavement and buildings to allow rainfall infiltration. In addition, pumping systems required to contain the

groundwater contaminant plumes under each decommissioning scenario were described. These "new" pumping systems represent what is required (number of wells, locations and pumping rates) to halt groundwater contaminant migration to the adjacent surface water bodies as a result of changes to the groundwater flow system created under each building teardown scenario. The study showed that between 7 and 12 pumping wells pumping between a combined rate of 16 to 30 gal/min would be required to halt contaminant flow to the Blue River and Indian Creek. Between five and 10 of these wells would be required in the Indian Creek Flow System alone (the system influenced by building footing tile drains). The current groundwater pumping system for the entire BFC consists of only 4 pumping wells, only one of which is located in the ICGFS. The required increase in the number of pumping wells to contain groundwater in the absence of building footing tile drains is significant.

3.4.2 Cattail Area

An area of water retention corresponding roughly to the former Blue River channel is located in the far northeast corner of BFC. This area, informally called the "Cattails", can be seen as a lightly shaded area in between wells 45 and 265 (Plate 1). The location of this cattail area in relation to the historical (pre-1970) location of the Blue River channel is shown in Figure 3.13. Fieldwork studies determined that recharge from this drainage area causes groundwater mounding which deflects groundwater to the east soon after entering the lower Northeast Area (DOE 2000). Organic rich environments such as exist in the cattail area are also conducive to contaminant degradation by biological means. If the cattail area did not exist, groundwater would flow unabated off of the BFC northeast to the Blue River. The cattails, in effect, serve to contain groundwater flow and direct it eastward toward existing pumping wells designed to capture groundwater and inhibit flow to the Blue River. This is further discussed in Section 3.11 below.

3.5 Velocity of Groundwater at the BFC

3.5.1 BRGFS

The horizontal hydraulic gradient in the BRGFS varies significantly due to topography. While most of the BFC has gentle sloping topography, over 30 ft. of elevation change occurs from the western side to the eastern side of this flow system. This change in surface elevation influences the potentiometric surfaces, and corresponding groundwater velocities as shown below.

In the northern portion of the flow system near wells 234L, 232L and 24L (Plate 1), groundwater velocities have been calculated to be several hundred feet per year with an eastward flow direction.

In the area north of Building 2306/2312 near wells 18L, 77L and 45L (an area that exhibits significant elevation change from west to east) flow rate ranges from 3000 -5000 ft/yr. with flow, on average, to the east-northeast.

Flow in the southeast portion of the BFC near wells KC95-03, KC95-01 and 42 produced a calculated flow velocity of 272 ft/yr in 2012 with a southeastwardly flow direction. See Table 3.6 - Southeast Parking lot pumping test.

3.5.2 ICGFS

Horizontal hydraulic gradients within this flow system are extremely variable as groundwater travels through an industrialized environment consisting of the MMB and surrounding buildings. Steep horizontal hydraulic gradients are present in the vicinity of man-made influences such as building footing tile drain systems and pumping wells. Conversely, extremely low horizontal hydraulic gradients are present in the vicinity of groundwater divides, such as the flow divide located in the central portion of the MMB (Figures 3.11 and 3.12).

Groundwater flow rates have historically been calculated for the northern and southern portions of the flow system. Manmade influences from pumping wells and building footing tile drains located in the central, western and eastern portions of the flow system distort groundwater flow in these areas so as to render flow velocities of little practical use. Therefore, flow rates are typically calculated using wells that show minimal effect from manmade influences.

The estimated groundwater velocity in the northern portion of the ICGFS is calculated to be several hundred feet per year with flow to the south. Velocities in the southern portion of the flow system running from the southeast parking lot (SEPL) to Indian Creek range from 1000-2500 ft /year with flow south-southeastwardly (Plate 1) .

3.6 BFC Hydrological Conceptual Model

A hydrological conceptual model is a description of how, where, and in what quantities water enters the groundwater flow system and the factors controlling groundwater movement between inflow and outflow locations. The conceptual model is derived from site-specific data and is intended to assimilate and condense concepts and ideas about groundwater flow into a series of statements that will aid in understanding factors influencing flow. The text below constitutes the BFC hydrological conceptual model as discussed in Laase (2010b).

Hydrostratigraphy:

- The KCP hydrostratigraphy consists of an upper clayey-silt and a basal gravel.
- The upper clayey-silt is approximately 20 feet thick and is relatively uniform in appearance.
- The basal gravel consisting of gravel sized rock in a clayey silt matrix is approximately one to eight feet thick. The unit is more variable in appearance than the upper clayey-silt.
- Relatively low permeability sandstone and impermeable shale underlies the alluvium at the BFC.

Recharge:

- The largest source of recharge at the BFC is leakage from underground water supply and fire protection lines with leakage concentrated in the heavily industrialized portion of the BFC.
- Rainfall recharge occurs on grassy areas and other areas receiving directed runoff such as the lower northeast area cattail region of the BFC.
- Groundwater throughflow from the Limestone Bluff on the north side of the BFC contributes a small amount of recharge to the alluvial aquifer.

Groundwater discharge is as follows:

- Groundwater in the BFC industrial areas primarily discharges to building footing tile drains.
- Groundwater not captured by the building footing tile drains discharges to the surrounding surface water bodies (Blue River and Indian Creek).
- A relatively small volume of groundwater discharges to the 001 stormwater Outfall.
- Groundwater is extracted by a series of pumping wells, treated at an on-site groundwater treatment facility and is discharged to the sanitary sewer by Permit.

With respect to hydraulic conductivity:

- Typical upper clayey-silt horizontal hydraulic conductivity ranges from 0.1 ft/d to 10 ft/d.
- Typical basal gravel horizontal hydraulic conductivities range from 1 ft/d to 100 ft/d.
- Occasionally BFC horizontal hydraulic conductivities exceed 100 ft/d.
- The highest measured horizontal hydraulic conductivities are found in the lower Northeast Area and Southeast Parking Lot (SEPL).
- Vertical hydraulic conductivities have not been measured but are assumed to be approximately one-tenth of horizontal hydraulic conductivity.

Finally, with respect to the BFC groundwater mass balance:

- Estimated cumulative groundwater recharge/inflow ranges between 141 and 250 gal/min.
- Estimated cumulative groundwater discharge is 127 gal/min.
- Total flow (in and out) of the BFC groundwater system is approximately 150 gal/min.

3.7 BFC Contaminants

The primary contaminants in soil at the BFC are chlorinated solvents, petroleum hydrocarbons and polychlorinated biphenyls (PCBs). The primary groundwater contaminant at the KCP is the solvent trichloroethylene (TCE) and it's anaerobically produced degradation products 1,2-dichloroethylene (1,2-DCE) and chloroethene (vinyl chloride). Other chlorinated compounds including 1,1-dichloroethane (1,1-DCA) and 1,1-dichloroethene (1,1-DCE) are found at the site but at significantly lower frequency and concentration than the three primary compounds.

PCBs have been detected in groundwater samples in wells located in areas of known PCB soil contamination. However, based on work performed in the 95th Terrace RFI, there is no evidence that PCBs, either colloidal or dissolved, are mobile at the BFC under natural flow conditions at concentrations above the site PCB groundwater clean-up standard (DOE, 1999). PCB detections in groundwater are related to PCB containing soil particles which are mobilized as a part of the sampling process.

The density and porosity of the aquifer material are the dominant properties of the transport medium that influence contaminant migration (Schwarzenbach et al. 1983). In addition, adsorption, oxidation, reduction, volatilization, biodegradation, and complexation reactions are also influenced by the aquifer properties which, in turn, further influence the mobility of organic contaminants in the subsurface (Mackay et al. 1985). The moisture content of the soils also affects the migration of a contaminant (Schwille, 1988).

The natural conditions within the groundwater at the BFC reflect the clayey-silt nature of the alluvial environment. Thus, the groundwater is low in oxygen which

mandates a number of other characteristics such as high iron and manganese. In general, groundwater in the alluvium has an alkalinity of ~200 mg/L, pH ~6.5 and dissolved oxygen < 1 mg/L. The defining natural characteristic of the alluvial aquifer, however, is the high arsenic content.

The fact that the alluvial aquifer at the BFC contains naturally-occurring arsenic at concentrations exceeding the primary drinking water standard was demonstrated some time ago and presented in technical journal articles (Korte and Fernando 1991, Korte 1991). In the ensuing years, however, realization of arsenic's toxicity and the ubiquity and danger associated with naturally-occurring arsenic have received greater publicity and emphasis. Shortly after the problem at the BFC was noted, two books were published (Nriagu 1994 and Chappell et al. 1994) discussing arsenic cycling in the environment, but it was the recognition of a large-scale public health crisis from arsenic in drinking water in Bangladesh that has heightened world-wide awareness (Kinniburgh et al. 2003). Indeed, since the original description of arsenic in the BFC aquifer, the U.S. lowered the drinking water standard from 50 µg/L to 10 µg/L. Moreover, research on arsenic has continued and expanded as demonstrated by a recent American Chemical Society Symposium Series (O'Day et al. 2005) that includes additional study of the BFC alluvial aquifer (Saunders et al. 2005). The latter study also documents the arsenic content of portions of the BFC alluvial aquifer at more than an order of magnitude above the drinking water standard. This investigation examined geochemical and microbial conditions at the BFC and compared those to another site in the US and with research reported worldwide. Apparently, similar natural processes are responsible for the public health disaster in Bangladesh as well as the naturally-occurring arsenic at the BFC. Unfortunately, arsenic removal from groundwater has proven both difficult and expensive. The recent book edited by O'Day et al. (2005) contains eleven articles concerning arsenic treatment demonstrating the inadequacy of currently-available methods. Hence, the alluvial aquifer at the BFC has no conceivable beneficial uses. Indeed, surface treatment of the organic contaminants is a much more addressable technical problem than treatment of the

naturally-occurring arsenic. In conjunction with the high concentrations of iron and manganese, there is no conceivable beneficial use of the alluvial aquifer at the BFC nor is complete restoration technically or financially feasible.

The narrative below discusses anthropogenic contaminants at the BFC but it is emphasized that the BFC aquifer is non-potable because of its natural characteristics of high arsenic, iron, manganese and low yield. Discharge of chlorinated contaminants to the Blue River and Indian Creek, therefore, is the principal regulatory concern regarding the contaminated aquifer.

Table 3.7 lists the physical and chemical characteristics of BFC groundwater contaminants including density, solubility, vapor pressure, octanol-water partitioning coefficient, and Henry's law coefficient. Information for Table 3.7 and the following discussion has been taken from Verschueren (1983), NIOSH (1985), The Merck Index (Budavari 1989), Howard (1989 and 1990), and Montgomery and Welkom (1990).

Field measurements taken during sampling at the BFC indicate that groundwater temperature ranges from 14.3° to 20°C. The higher temperatures are believed to be the result of leaking underground utility lines. The groundwater contaminants are liquids at these temperatures except for vinyl chloride, which is a gas. Most of the BFC contaminants have high vapor pressures (> 0.01), low octanol-water partition coefficients (< 2.7), and high Henry's law coefficients (> 100 atm). When released into the subsurface, these compounds tend to be mobile in soil and groundwater. In contrast, when released into surface water, these compounds tend to be removed quickly due to volatilization as supported by the relatively high Henry's law coefficients and relatively high vapor pressures.

PCBs are an exception to the generalizations of the previous paragraph. The chemical and physical properties of PCBs vary depending on the number and placement of chlorine atoms around the biphenyl molecule (Erickson 1986). For

example, Aroclor 1254, which has the greater quantity of highly-chlorinated congeners, is more viscous than Aroclor 1242—the latter being the mixture most widely used at the KCP (Moore and Ramamoorthy 1984).

In general, PCBs are thermodynamically stable and difficult to degrade. These compounds have very low vapor pressures (0.00006 to 0.001 mm Hg), low solubilities (0.05 to 0.2 mg/L (3 20°C) and moderate to high Henry's law coefficients (31 to 150 atm). When released into the subsurface, these compounds tend to have low mobility in soil and groundwater and a strong affinity for soil (EPA 1979). When released into surface or groundwater, these compounds tend to be removed through sorption onto sediment. Historically, PCBs were often described as insoluble or immobile. These terms, however, are not strictly true as PCBs dissolve and degrade slowly in the environment. Furthermore, most degradation pathways result in formation of more volatile and more soluble compounds. Hence, an old, degraded spill may show a greater percentage of higher chlorinated compounds in the source area as compared to the original mixture spilled. Similarly, PCBs that have migrated from an old spill will typically show greater percentages of the less-chlorinated congeners relative to the original mixture spilled.

Many of the groundwater contaminants listed in Table 3.7 are either minor contaminants or likely laboratory artifacts. For example, acetone, chloroform, methylene chloride, 2-butanone, and 4-methyl-2-pentanone are usually considered to be laboratory contaminants. These have never been reported consistently at any location at the BFC. Therefore, the remainder of this section emphasizes the fate and transport characteristics of the principal groundwater contaminants found at the BFC (TCE and its degradation products - 1,2-DCE, and vinyl chloride). Their fate and transport characteristics will encompass those of the remainder of the volatile organics listed in the Permit (Tables 3.8 and 3.9). Focusing on the three primary compounds is appropriate because their presence defines the extent of groundwater contamination, these are typically present in the highest

concentrations, they have fate and transport characteristics that are similar to the minor contaminants, and include the lowest MCLs (5 µg/L for TCE, 70 µg/L for 1,2-DCE, and 2 µg/L for vinyl chloride).

3.8 Chlorinated Solvents – Behavior in the BFC Subsurface Environment

3.8.1 TCE

TCE biodegrades under anaerobic conditions (like that at the BFC) principally into cis-1,2-DCE (some trans- 1,2-DCE may also be formed) and then into vinyl chloride (Kloepfer et al. 1985; Wilson and Wilson 1985; Cline and Viste 1985; Barrio-Lage et al. 1986). The minor presence of 1,1-DCE at the BFC is also attributed to degradation of TCE. Although prevalent as groundwater contaminants, there is no evidence that 1, 2-DCE and vinyl chloride were used or disposed of at the BFC (DOE 1986a; Benkovich and Jackson 1987). TCE, however, was heavily used. 1, 2-DCE and vinyl chloride behave somewhat differently than TCE. These differences can be described by the K_{ow} of the compounds. K_{ow} is a function of the water solubility and sorptive capacity of an organic compound onto solid organic matter (Table 3.7). The log K_{ow} , values for the two degradation products are smaller than the parent compound: TCE = 2.29; 1,2-DCE = 1.86; and vinyl chloride = 0.6 (EPA 1979). Thus, vinyl chloride is considered a highly mobile compound, while 1,2-DCE is less mobile than vinyl chloride but more mobile than TCE. Biodegradation rates of chlorinated solvents are difficult to predict or measure because they vary considerably with soil type, water chemistry, hydrologic conditions, microbes present, organic matter content, temperature, pH, Eh, oxygen content, and the presence of nutrients. For example, in a like manner to TCE, 1,1,1-TCA undergoes biologically mediated reductive dechlorination and is apparently responsible for the minor contaminant 1,1-DCA. The biodegradation of 1,1,1-TCA (Vogel and McCarthy 1985) was originally reported as slow (Cline and Viste 1985), relatively rapid (Arvin 1989), or non-existent (Wilson et al. 1989). It was eventually recognized that varying results

were the consequence of different experimental conditions, such as flow rates and reaction times, used by various researchers (Bae et al. 1989). 1,1-DCE is also formed from 1,1,1-TCA but by abiotic reductive dehalogenation (Curtis and Reinhard 1989).

The foregoing discussion assumes the only relevant process within the aquifer is reductive dechlorination. However, microbial mineralization to carbon dioxide can be an important component of in-situ vinyl chloride and 1, 2-DCE natural attenuation under oxic conditions and, in certain environments, in the absence of detectable oxygen. (This process is considered in detail in Section 3.12.2 for the aquifer/surface water interface). In contrast to reductive dechlorination, the potential for mineralization of chloroethenes increases with decreasing chlorine number (McCarty and Semprini, 1994; Bradley and Chapelle, 1996; Bradley, 2003) and the end product(s) of microbial mineralization of chloroethenes, primarily carbon dioxide (or carbon dioxide and methane under methanogenic conditions), however, are not unique to chloroethene mineralization and, consequently, are not diagnostic of this process and are usually neglected. Thus, ignoring mineralization and assuming that reductive dechlorination is the only degradation mechanism underestimates the actual overall contaminant degradation.

3.8.1.1 History of TCE/Use at the BFC

TCE use at the BFC was as a cleaner and degreaser. TCE use, specifically in manufacturing operations for the production of aircraft engines and subsequently through the manufacture of non-nuclear components of nuclear weapons, cannot be underestimated.

Trichloroethylene has played an important role in the industrialization of this country. However, the prevalent use of TCE has also left its fingerprint upon our environment making it a common soil and groundwater contaminant.

TCE has been a widely used solvent in the past, especially for industrial metal cleaning and vapor degreasing purposes. TCE was discovered in Germany in the mid-1800s; however, it was not until the early 1900s before industrial TCE production began in Europe. The early use for TCE was in laundries, as well as for textiles and varnishes.

Manufacturing of TCE did not begin in the United States until the early 1920s, following the end of World War I. Although production of TCE in the United States began in the early 1920s, significant industrial use of the solvent did not begin until the late 1920s. The most common uses in the United States for TCE early in its production life was for polishes, degreasing in the metals industry and for fats extraction in the food industry.

TCE use began to significantly increase throughout the metal fabrication industries as improvements were made in vapor, spray and immersion degreasing equipment. (Dewitt, 2013)

With the outbreak of World War II in the early 1940s, the major use of TCE was as a metal degreaser to support the war effort. Major manufacturers of TCE increased production from approximately 50 million pounds per year at the start of the war to approximately 200 million pounds per year at the end of the war in 1945. By the end of the 1940s, total TCE production had reached approximately 225 million pounds per year.

In the 1950s, total TCE production reached approximately 350 million pounds by the end of decade. Reasons for the increase of TCE use in the 1950s were the Korean War and the ever increasing industrialization of the United States. Throughout the 1950s, in excess of 90% of all TCE production in the United States was used for metal degreasing purposes.

During the 1960s, TCE continued to be the most popular vapor degreasing solvent due to both its price and performance. These facts led to an increase in domestic production and its ultimate peak production in the United States by the end of the decade and 1970 at approximately 600 million pounds per year.

After reaching peak production in 1970, the use of TCE in the United States began to decline. This decline was due to both environmental and economic factors. The 1970 Clean Air Act put emission limits on TCE due to its contribution to ozone and smog creation. Then in 1975, the National Cancer Institute found that TCE could cause tumors in mice and in 1976 TCE was put on the Hazardous Substance List by the United States Environmental Protection Agency.

Economic pressures lead to a decline of TCE use in the 1970s. The price of TCE precursor production chemicals increased thus leading to the shutdown of several major production facilities. These plant shutdowns, in turn, lead to a TCE price doubling into the mid-1970s. Other companies were not inclined to get into the TCE production market due to both the economic and environmental conditions. As a result, TCE production fell to approximately 400 million pounds per year by the end of the decade.

TCE production and use continued to fall during the 1980s thru the 2000s as increased environmental regulations took hold. In 1980, TCE was regulated as a hazardous waste material. Additional regulation occurred under the Safe Drinking Water Act in 1985 with an establishment of a maximum contaminant level (MCL) of five (5) parts per billion and the lowering of the Permissible Exposure Level by the Occupational Safety and Health Administration in 1989. Although during the 1990s TCE had been suggested as a replacement for other solvents banned under the 1990 Clean Air Act Amendment, the annual United States' production of TCE had fallen to approximately 100 million pounds per year in 2000 and continues to fall through present day.

The importance of TCE to the industries of this country cannot be understated since TCE was considered the best degreasing solvent available and, in effect, helped build America. At the same time, its popularity and use has also given the United States a lasting environmental legacy of soil and groundwater impacts. TCE is a common contaminant which has been detected at approximately 60% of all National Priority List sites and countless other properties throughout the United States. The majority of environmental impacts of TCE have occurred from the use of TCE as a metal degreaser, due primarily to the releases of spent TCE prior to the advent of environmental regulations in the 1980s.

3.8.2 Aromatic Compounds

Benzene and other aromatic compounds are found sporadically in BFC groundwater— most regularly at the site of a former gasoline filling station near the southeast corner of the MMB (see data for well KC92-187U). Although fuels and oils may consist of hundreds of compounds, it is not unusual that other fuel-related compounds are not detected in BFC groundwater. Aromatic compounds are generally of chief interest in a fuel spill because they make up over 90% of the water-soluble fraction, even though they may account for less than 50% by weight (Coleman et al. 1984). In addition, many of the other components of a fuel spill are much more biodegradable (Atlas 1981). For this reason, aromatic compounds such as benzene often remain when other vestiges of a fuel spill have been biodegraded. Thus, biodegradation is considered to be an important removal mechanism for petroleum hydrocarbons at the BFC because little evidence of groundwater contamination by petroleum hydrocarbons has been found despite evidence of relatively large disposal of petroleum wastes. The sporadic groundwater contamination indicates that the more mobile, typically more easily degradable petroleum compounds are being removed by subsurface microbes. In other words, groundwater flow rates at the BFC are slow enough relative to microbial removal processes that these contaminants cannot migrate in the

groundwater. The research literature strongly supports this concept of natural biological attenuation of petroleum hydrocarbons (Davis et al. 1994, McAllister and Chiang 1994; Thierrin et al. 1993).

3.9 Conceptual Site model Groundwater Contamination

When groundwater contamination with chlorinated solvents like TCE was initially investigated the prevailing thought in the scientific community was that pumping contaminated groundwater from an aquifer and treating it before offsite disposal would eventually eliminate all groundwater contamination through the physical removal of the contaminant from the aquifer (EPA, 1996). Subsequent work demonstrated that chlorinated solvents may form pools of pure phase product. These Dense Non Aqueous Phase Liquids (DNAPL) (in the case of solvents denser than water like TCE), could be sources of contamination for decades and possibly centuries (Wilson and Conrad 1984; Schwille 1988). More recent work has shown that the idealized pools mentioned above represent an oversimplification of DNAPL behavior for a site such as the BFC. The alluvium at the BFC is two to four orders of magnitude less-permeable than that envisioned in the idealized "pool" scenario.

The general theory of DNAPL movement has been provided by Schwille (1988). DNAPL spilled or leaked to the ground above the water table moves under the force of gravity to the capillary fringe (Figure 3.14). At first, surface tension between the water and the DNAPL is sufficient to prevent penetration of the water table. As more DNAPL is added, eventually enough reaches the water table to overcome the water capillary pressure at the capillary fringe, and the solvent penetrates the water table. Under the force of gravity, the solvent migrates rapidly until it encounters a layer of lower permeability. This scenario was suspected at some of the sources at the BFC.

A review of the literature, however, shows that the idealized circumstances presented in Figure 3.14 present an oversimplification for a site such as the BFC. Figure 3.14 is based on models using glass beads (Figure 3.15), which can only simulate the circumstances for very coarse soil conditions. In contrast, the alluvium at the BFC is two to four orders of magnitude less-permeable. The actual distribution of DNAPL in such a low-permeable system requires additional consideration. Figure 3.16 shows the heterogeneous nature of DNAPL distribution (at the microscale) such as might be encountered in subsurface soils. The system depicted in the figure is coarser than what is present at the BFC but it illustrates how DNAPL is trapped as ganglia in discontinuous phases. Indeed, in a finer-grained system where the capillary entry pressure is significantly higher, a "pool" such as is shown in Figure 3.14, cannot form and the overall distribution will look more like that shown in Figure 3.16. This figure presents a DNAPL distribution developed for a fine grained system in the laboratory (Wilson et al. 1989; Conrad et al. 1988; Conrad et al. 1992).

A field-scale conceptual drawing of such a distribution is shown in Figure 3.17. The figures show that "pools" are not formed and that DNAPL distribution is highly discontinuous. The figure suggests that, especially in heterogeneous field systems, there are dead-end and even closed off pores where water or DNAPL can accumulate and be virtually impossible to displace. In a sense, the circumstances can be thought of as resulting in many "micropools" of DNAPL interspersed with some water-filled pores as well. In this case, discrete sampling over a 5-ft zone, is unlikely to show large variations in contaminant concentrations. DNAPL ganglia will be dispersed throughout the zone, and even though films of water surrounding the ganglia may contain saturated concentrations of the solvent, bulk samples, even from a very small area, will show significantly lower concentrations.

The model depicting idealized pools was replaced to more clearly show the conditions believed to exist, that of discontinuous ganglia that will be sources of

contamination to the groundwater for a considerable period of time. It is also concluded, because of the likely presence of DNAPL and the tortuous flow path for groundwater at the KCP, that pump-and-treat is unlikely to significantly alter groundwater contaminant concentrations for an extensive period of time.

3.9.1 Fit of the model to the KCP

DNAPL at the BFC is composed of TCE, with an aqueous solubility of 1,100 mg/l. One area of DNAPL is located in the area of the former plating building degreaser just east of MMB. Immediately downgradient of a zone of residual DNAPL, one would expect the local groundwater concentration to be approximately 1,100 mg/l (well 192 exhibited TCE concentrations in this range prior to startup of adjacent pumping well 276). As one moves downgradient, TCE concentrations will be less due to hydrodynamic dispersion. Dispersion in the subsurface results in a lowering of concentrations along the axis of a plume in the downgradient direction. The maximum concentration of 1,100 mg/l can only be observed immediately adjacent to the DNAPL and will not be observed anywhere down-gradient of the source zone. This is the contaminant distribution noted in the area of the former vapor degreaser. Besides hydrodynamic dispersion, other factors including specific monitoring well placement and the amount of contaminant degradation over time (biotic or abiotic) work to produce lowered contaminant concentrations in wells downgradient of DNAPL source zones.

3.10 Potential for Contaminant Migration in Bedrock

As noted previously, the Knobtown Sandstone bedrock subcrops across the central portion of the BFC. Detailed lithologic descriptions of the bedrock can be found in Korte et al. (1985), Madril et al (1986), and in DOE (1990a). Several bedrock wells were installed to determine the local stratigraphy, conduct permeability tests, and monitor groundwater quality in the bedrock aquifer. When the site-characterization program was initiated in 1983, it was suspected that any

dense, chlorinated-solvent contamination, if present in bedrock, could migrate, by means of density-driven flow, in the direction opposite of groundwater flow if the bedrock surface dipped in the opposite direction. This was apparently a possibility at the BFC, where groundwater flow was toward the surface-water system (Blue River) and the bedrock dipped in the opposite direction (Figure 3.5).

Consequently, a rigorous program was initiated to determine the extent of contaminant migration in the bedrock. This program was complicated by the difficulty of obtaining reliable water samples from the Knobtown Sandstone. To obtain a water sample from the bedrock only and not from the overlying alluvium, standard engineering practices dictate that steel surface casing be cemented into the bedrock to a depth of 5 ft. This seal prevents alluvial water from directly entering the well and biasing the results. However, at the BFC, the Knobtown Sandstone is only 10 ft thick at a maximum and much less in areas where erosion has removed a substantial portion. If the wells were located downdip of suspected contaminant source areas where substantial thicknesses of Knobtown Sandstone are located or where the Pleasanton Shale separates the sandstone from the overlying alluvium, the migrating contaminants would take thousands of years to reach them. To address this problem, a worst-case, contaminant-transport simulation using a computer model was employed to predict the extent of bedrock contamination. This simulation demonstrated that an immiscible chlorinated-solvent plume in the Knobtown Sandstone would migrate less than 200 m (656 ft) in a thousand years (Madril et al. 1986). The model assumed that no biodegradation or retardation would take place. Simulations of groundwater transport were not performed on the Pleasanton Shale because packer tests demonstrated that the hydraulic conductivity was too small to measure.

After completing this work, two events occurred that required re-evaluation of the bedrock-characterization program. First, regulatory compliance required further characterization of the bedrock system, including the installation of additional wells 1600 ft (500 m) downdip from the Underground Tank Farm area. Second, review of the technical literature indicated that chlorinated solvents do not

migrate substantial distances in the immiscible phase. The first event resulted in the installation of three wells 500 m downdip from the Underground Tank Farm area (wells 122, 123, and 124 on Plate 1). These bedrock monitoring wells are uncontaminated.

A brief field program was conducted in 1992 to further examine the premise that bedrock at the BFC provides a barrier to the downward migration of contaminants. Prior conclusions were based on permeability testing, examination of cores, groundwater sampling, and a literature review. In late 1992, field investigations at the KCP provided the opportunity to collect samples of bedrock below suspected pools of DNAPLs, such as PCBs and TCE (DOE 1993c, 1993d). Two coreholes were drilled in appropriate locations, and samples were collected from the alluvium, the alluvium-bedrock interface, and competent bedrock. The bedrock sampled was Knobtown Sandstone, the more permeable of the bedrock formations at the BFC (DOE 1992d). The investigation was undertaken to determine whether there was penetration of the bedrock by DNAPLs that may have collected at the bedrock/alluvial contact. Performing a determination such as this is inherently imprecise because the bedrock-alluvial contact is gradational due to weathering of the bedrock. In other words, prior to burial by the alluvium, the Knobtown Sandstone was exposed to wind and water erosion and now grades from the alluvium through weathered zones to competent, undisturbed bedrock. Precautions were to be taken to collect samples that encompassed the bedrock interface, yet the alluvium also had to be carefully sealed from the bedrock to ensure there was no conduit for downward migration of contaminants.

The investigation showed high contaminant concentrations in the alluvium and the weathered portion of the bedrock. For example, a sample from one of the coreholes contained 290,000 µg/kg TCE. The other corehole contained 1300 mg/kg of PCBs at or near the alluvial-bedrock interface. No PCB contamination was found in subsequent bedrock samples from this corehole. The lack of contamination in bedrock samples demonstrated that the bedrock provides a

barrier to downward migration of contaminants. The results of this study further substantiated the previous conclusion that the potential for contaminants to migrate into bedrock is insignificant (DOE 1992d).

Sampling and analysis of bedrock groundwater (principally wells 27, 122, 123 and 124) was continued on a quarterly basis for several years. There were a few spurious VOC detections in the early years of monitoring but contamination was never confirmed in subsequent sampling and was later found to be related to the type of equipment used in sampling of these well types. Ultimately, with regulatory approval, bedrock monitoring wells 122 and 123 are now sampled annually. Contamination is never detected.

In summary, water quality and hydrological data obtained from the bedrock wells support the following conclusions:

1. The contamination in the alluvium is highly unlikely to contaminate the bedrock.
2. Groundwater flow rates in the bedrock are so slow that any potential contamination would have no effect on potential off-site receptors.
3. The groundwater yield of the Knobtown Sandstone is so low that it is unsuitable for any beneficial use in the vicinity of the plant.

3.11 Application of Natural Attenuation Protocol

Previous work evaluated natural attenuation by biodegradation at the BFC based on a widely-used protocol developed for the Air Force Center for Environmental Excellence (AFCEE/Wiedemeier et al. 1996). Based on samples collected from the ICGFS near the MMB, the aquifer received a relatively low score and has “limited evidence for biodegradation of chlorinated organics” according to this protocol (ORNL 1998). Biodegradation is occurring, however, because the original spills of TCE have been transformed to 1, 2-DCE and VC downgradient from the source areas. Moreover, subsequent data collection suggests biodegradation is so rapid in certain locations that a repeat application of the

natural attenuation protocol in these regions would show significant biodegradation.

For example, rapid biodegradation is apparently demonstrated in the so-called “cattail area” located in the Northeast Area (Plate 1). Potentiometric surfaces indicate significant contamination (up to 1000 µg/L of 1,2-DCE) migrates toward Boone Creek through this cattail zone. Contamination, however, is absent in the creek itself and in adjacent wells. Based on flow rates as used in the modeling performed by Laase (2010), approximately 125 days are required for groundwater to flow across the cattail region indicating 1000 µg/L of 1,2-DCE must be degraded in this same time period. Besides the absence of contamination, high biodegradation rates are supported by the presence of a dark, apparently organic zone found in the shallow subsurface in this location. Although, this “cattail area” has not been subjected to appropriate field testing to determine the nature and distribution of organic matter, contamination, or redox conditions, a variety of field and laboratory studies have reported biodegradation rates rapid enough to explain the removal of 1, 2- DCE in this zone. For instance, Broholm and co-workers (2005) reported 50% 1, 2-DCE removal in 204 days increasing to 90% in some cases with addition of methane. Similar data have been reported by Bradley and Chapelle (1997) who reported average degradation rates for 1, 2-DCE in stream bed sediments of 0.6 %/day with an upper limit of 0.8%/day. The latter rate would eliminate 1, 2-DCE in 125 days. Klier and others (1999) reported greater than 95% mineralization of cis-1, 2-DCE within 70 days using soil micro-organisms while Bradley and Chapelle (1998), in a subsequent study, reported 15 to 100% removal after an 8-day incubation period.

A calculation of contaminant mass at two different times cannot be performed for the BFC because of complicating factors of multiple sources, contaminant removal by pumping, and leaking utilities. Similarly, contaminant trend graphs for individual monitoring wells cannot be used as evidence of plume stability because pumping and treating for more than 20 years have affected the

concentrations. That concentrations have decreased, however, is shown by many of the monitoring well trend graphs. Figures 3.18 and 3.19 show that wells 69U and 39U show decreases in concentration for one or more of the major contaminants. Well 37 shows a large decrease in TCE which is apparently degrading and maintaining the 1, 2-DCE mass. Well 37 may have been affected by a soil mixing treatability study and by nearby pumping wells but well 115 should not have been affected by any factor other than time. Unfortunately, in neither case, are there nearby wells certain to be free of influence of remediation or construction to estimate a decrease in plume mass based solely on in-situ, natural processes. Additional support for contaminant removal by biodegradation at the BFC is derived from work reported for Dover AFB, Delaware (Everett et al. 2006, Kennedy et al. 2006). In this work, common sulfate-reducing soil bacteria were stimulated with injections of Epson salt ($\text{MgSO}_4 \cdot 7\text{H}_2\text{O}$) and sodium (L) lactate ($\text{NaC}_3\text{H}_5\text{O}_3$) (Epson salt provides the sulfate and the lactate is the organic matter), thereby facilitating the geochemical conversion of native iron minerals into iron sulfides. Iron sulfides chemically reduce many common chlorinated compounds including PCE, TCE, and 1, 2-DCE. As these authors noted, iron sulfides reduce chlorinated hydrocarbons at many sites where sulfate and organics occur, but the process has been overlooked because (1) investigators were not looking for it and (2) no distinct daughter products are generated. There is significant evidence these processes are occurring at the BFC. Saunders et al (2005) collected a series of groundwater samples to examine the mechanism for arsenic dissolution at the BFC and reported the involvement of both sulfur and iron-reducing indigenous bacteria—thereby verifying existence of the same geochemical process requiring the same type of bacteria as for the anaerobic chlorinated solvent removal described above. In addition, a deep-soil mixing technology demonstration treatability study performed in the BRGFS reported organic matter in BFC subsoil at concentrations of approximately 0.5%. Although not a high value, these data demonstrate there is sufficient organic substrate present in the deeper subsurface to support microbiological processes that remove chlorinated solvents. Additional evidence is provided by microcosm studies

recently performed for the aquifer/surface water interface and described in Section 3.12.

3.12 Natural Attenuation at the Groundwater/Surface Water (Indian Creek/Blue River) Interface

Contaminant degradation by microbes in the groundwater/surface water interface is a potentially robust mechanism for removing low level contaminants before they reach the surface water (Blue River/Indian Creek). Research has shown that when an anaerobic groundwater plume (as at the BFC) discharges to surface water, indigenous microbes within this interface rapidly oxidize 1, 2-DCE and vinyl chloride (Bradley and Chapelle 1998) representing a “...non-engineered opportunity for efficient contaminant bioremediation” (Figure 3.20).

The fact that dilution and/or degradation at the aquifer/surface water interface are preventing any surface water detection of contaminants is supported by data from the BRGFS. A brief seepage study (DOE 1994a) showed that contaminants were, indeed, present at the riverside and at least three feet below the riverbed. Nevertheless, contaminants have never been confirmed in the river. The lack of site contaminants in the river is supported by more than twenty years of surface water monitoring as documented in the BFC annual Groundwater Corrective Action Reports (DOE, 2012).

3.12.1 Surface Water/ Groundwater Interaction

The BRGFS contaminants discharge directly to the Blue River. This is not true for the ICGFS because utilities associated with the buildings have inadvertently constrained plume growth. The latter effect may not be permanent because effects of future construction, repairs and operational changes cannot be predicted. Data collected to date, however, suggests an ICGFS/Indian Creek interaction will

mimic what is observed at Blue River. For example, although the Blue River watershed is approximately three times the size of that of Indian Creek, low flow discharges, the most critical period for addressing groundwater contaminant effects, are similar. Under non-pumping conditions, the Blue River presumably undergoes periods when it is either a gaining or losing stream depending on precipitation, but pre-pumping maps of the potentiometric surface indicated the stream was gaining on all occasions measured. Similarly, Indian Creek is usually a gaining stream in the reach alongside the BFC as well, although that conclusion is tempered because there are so few creek-side monitoring wells.

Well logs from drilling performed on both sides of the Blue River show there is less than three feet from the bedrock to the surveyed stream channel bottom (Figure 3.21). The section of stream where the plume discharges has been channelized and the bottom consists of a silt-clay layer over gravel/bedrock and debris. The quantity of organic matter in the sediments is unknown but may govern whether microbial processes are aerobic or anaerobic. Although there is no control from well logs, stream inspections and surveys suggest conditions are similar for Indian Creek alongside the BFC.

As the aquifer approaches Blue River, well logs indicate a shift from a highly anaerobic system to one that is less so. For example, the well log for KC85-35, drilled in the center of the BFC, indicates a uniform, dark-gray, silty-clay but well KC89-121, near the river, has a more oxidized appearance including limonite stains (Figure 3.1). Once again, such data do not exist for the ICGFS, but there is no reason to believe conditions are not similar.

With respect to natural groundwater quality, the highest iron, manganese and arsenic natural occurrences, because conditions are becoming less anaerobic, are not found in wells near the streams (Figure 3.22). There are no DO measurements in the water from these wells but measured DO (Ryon et al. 2000) in the Blue River and Indian Creek (>7 mg/L) is comparable to measured DO at sites where

aerobic degradation at the interface was favorable (see below).

Initial studies (1980s and early 1990s) of spills of chlorinated ethenes such as TCE or tetrachloroethene (PCE) reported the latter compounds were not transformed under aerobic conditions, but were degraded anaerobically, principally to 1, 2-DCE and vinyl chloride. The presumption was, therefore, that TCE spills in aerobic environments underwent no degradation while those in anaerobic environments produced an even more toxic compound, vinyl chloride, which was then persistent (Ward 1985). Consequently, sequential anaerobic/aerobic degradation of chlorinated solvents in groundwater was proposed as a means of removing all of the contaminants (Barrio-Lage et al. 1986).

Although easily performed in the laboratory, aquifer manipulation on this scale has typically proven intractable for field application in clayey-silt soils such as at the BFC. Subsequent research demonstrated the presumed recalcitrance of vinyl chloride under anaerobic conditions was incorrect. Such results were not surprising, because it had already been demonstrated that PCBs, chlorinated compounds much more difficult to degrade, were removed, albeit slowly, under anaerobic conditions (Rhee et al. 1989). Specifically, vinyl chloride anaerobic degradation was demonstrated both in iron reducing systems (the BFC is an iron-reducing system) (Bradley and Chapelle 1996, 1997, Bradley et al. 1998a) and in systems high in humic acids (Bradley et al. 1998b).

Similarly, Chen et al. (1996) demonstrated complete mineralization of 1, 1, 2, 2-tetrachloroethane under anaerobic conditions. As was summarized previously, recent research indicates 1, 2-DCE and vinyl chloride are being removed within the aquifer at the BFC—although, as groundwater monitoring and modeling demonstrates, not sufficiently to prevent migration to the Blue River and Indian Creek if the drains, pumps, and leaks that now constrain the plume are ever altered or repaired.

Not until 1998, was there published confirmation that discharge of an anaerobic plume to an aerobic surface water environment provided appropriate conditions for complete mineralization of chlorinated ethenes (Bradley and Chapelle 1998). In the latter study, the concentration of 1,2-DCE ranged from 1900 µg/L to 40 µg/L (the latter being 7m from the creek). Sampling of the creek bed revealed 8.9 µg/L at a depth of 20 cm in the creek bed, but contaminants were not detected in the upper 10 cm. Dissolved oxygen (DO) was approximately 2 mg/L in the shallow bed sediments and 6 mg/L in the overlying creek water. As of this writing, no data are available regarding the quantity of DO in Blue River or Indian Creek sediments, but the surface water concentrations have been reported as >7 mg/L (Ryon et al. 2000).

3.12.1.1 Contaminant Concentrations at the Aquifer/Surface Water Interface

The KCP reacted quickly to the discovery of groundwater contamination such that a pump and treat system was installed in 1989, less than three years after the first well was installed alongside the Blue River. Initially, maximum contaminant (chiefly 1, 2-DCE and vinyl chloride) concentrations at wells near the Blue River were somewhat more than 100 µg/L.

A few months after the pump and treat system was activated, contaminants were typically not detected at the riverside. Subsequently, the KCP installed a zero-valent iron treatment wall and the pump and treat system was turned off. Unfortunately, there was bypass flow such that contamination once again appeared in riverside wells. Once pump and treat was resumed, riverside contamination again decreased and now appears to be an isolated remnant with no more than 10 µg/L of total VOCs (DOE 2006a).

The only study to examine contamination in the bed sediments of the Blue River, the aforementioned seepage meter study (DOE 1994a), occurred in 1991 after the

pump and treat system was operating but before contamination at the riverside had been eliminated. Riverside concentrations had already dropped to approximately 10 µg/L in the monitoring wells when a series of seepage meters and mini-piezometers were installed (DOE 1994a).

Mini-piezometers 4 and 5 were installed in the river near the edge and mini-piezometer 6 was installed further in the river. Mini-piezometer 7 was installed alongside the river and showed contamination similar to a nearby monitoring well. Mini-piezometer samples were collected approximately 3 ft below ground surface. Seepage meters may under-predict actual VOC concentrations because they have a large headspace. Nonetheless, a seepage meter, located in the river, had 18 and then 6 µg/L of 1,2-DCE and 24 and 13 µg/L of vinyl chloride on April 4 and August 16, 1991 respectively. Mini-piezometers labeled 4, 5 and 6 all had contaminants at one time or another from the four times they were sampled—up to 52 µg/L 1, 2-DCE and 45 µg/L vinyl chloride. Finally, contamination observed in the riverbed was not observed in the river itself—a result of degradation within the bed sediments and dilution.

3.12.2 Relevant Field Measurements at the BFC

The success reported by Bradley and Chapelle and co-workers (1996, 1997, 1998a, 1998b) evaluating degradation at the aquifer/streambed interface led the DOE to undertake a series of microcosm experiments in order to quantify the processes for the Blue River and Indian Creek Flow Systems. The laboratory work was performed by the US Geological Survey (USGS), in cooperation with the DOE/KCP. The work consisted of an assessment of the potential for *cis*-DCE and VC biodegradation in the streambed sediments at the BFC. The results are summarized below and the complete report is presented at the end of this section. The specific focus of the investigation was the potential for biotic and abiotic *cis*-DCE and vinyl chloride degradation in surficial and underlying hyporheic zone (approximately 1 m below the sediment and water interface) sediment from the

Blue River and its tributaries, Indian Creek and Boone Creek. A series of carbon-14 (¹⁴C) radiotracer-based microcosm experiments was conducted to assess the mechanisms and products of degradation.

Studies were performed with nominal concentrations of 300 µg/L. Substantial degradation of [1,2-¹⁴C] *cis*-DCE and [1,2-¹⁴C] Vinyl chloride to ¹⁴C-carbon dioxide (¹⁴CO₂) was observed in all viable surficial sediment microcosms prepared under oxic conditions. No significant accumulation of reductive dechlorination products was observed. Such results indicate that microbial mineralization processes involving direct oxidation or cometabolic oxidation are the primary mechanisms of *cis*-DCE and vinyl chloride biodegradation in oxic stream sediments at the BFC.

Similarly, substantial mineralization of [1,2-¹⁴C] vinyl chloride was observed in all viable surficial sediment microcosms incubated in the absence of detectable oxygen (dissolved oxygen concentrations less than 25 micrograms per liter). In general, the accumulation of mineralization products (¹⁴CO₂ and ¹⁴C-methane (¹⁴CH₄)) predominated with only trace level detection of the reductive dechlorination product, ¹⁴C-ethene. In contrast, microbial degradation of [1,2-¹⁴C] *cis*-DCE by reductive dechlorination or mineralization was not significant in the absence of detectable oxygen. In addition, the potential for [1,2-¹⁴C] vinyl chloride biodegradation was significant in sediments from the deeper hyporheic zones under both oxic conditions and in the absence of detectable oxygen.

Taken together, the results support the hypothesis presented in section 3.12 above that microbial mineralization processes in streambed sediments at the BFC are an important component of *cis*-DCE and vinyl chloride degradation and help explain why chlorinated solvents have not been confirmed in the Blue River or Boone Creek – even though groundwater contamination is adjacent to the Blue River. This conclusion is specific to oxic conditions for both 1, 2-DCE and vinyl

chloride and for vinyl chloride even in the absence of detectable oxygen. These results also demonstrate that an evaluation of the efficiency of in situ *cis*-1, 2-DCE and vinyl chloride biodegradation in streambed sediments, based solely on observed accumulations of reduced daughter products, probably would underestimate substantially the total extent of contaminant biodegradation and, thus, the potential importance of the hyporheic zone and streambed sediments as barriers to the discharge of contaminated groundwater.

Based on the occurrence of substantial methanogenesis, reducing conditions prevailed in two of the three Blue River sediments evaluated as well as the Boone Creek microcosms. In contrast, little to no methane was detected in Indian Creek and in the third Blue River microcosms, which indicates that relatively oxidized conditions persisted in these sediments. Nonetheless, significant degradation of [1,2-¹⁴C] vinyl chloride was observed in all experimental microcosms prepared with sediment samples collected from the hyporheic zone at the BFC, irrespective of oxygen treatment (Figures 3.23 and 3.24). Vinyl chloride degradation occurred even in the absence of detectable oxygen and, in some cases, in the presence of substantial methanogenesis. Hence, microbial activity in the hyporheic zone may remove the traces of VC in the groundwater, even before the water contacts the surficial sediments. This finding indicates that the hyporheic zone provides a significant attenuation buffer for any vinyl chloride occurring in groundwater discharging to the stream systems at the BFC. These results re-emphasize the conclusion that an evaluation of in-situ vinyl chloride biodegradation in BFC stream sediments that is based solely on the accumulation of reduced daughter products may underestimate significantly the extent of contaminant biodegradation and the potential importance of the hyporheic zone as a contaminant attenuation buffer.

3.13 Specific Conceptual Models for the Origin and Movement of Contamination in the BRGFS and ICGFS

This section presents conceptual models for contaminant fate and transport in groundwater at the Bannister federal complex. The conceptual models are derived from DOE 1995a and consider the presence of dense non-aqueous-phase liquids (DNAPL) in the alluvial aquifer at the BFC, based on field data and the technical literature (Korte et al. 1995). EPA (1993a) concluded that up to 60% of National Priorities List sites may have DNAPL contamination in the subsurface; a significant percentage of RCRA corrective action facilities are also thought to be contaminated with DNAPL.

3.13.1 BRGFS

Developing a conceptual model for contaminant migration in the BRGFS is complicated because the activities that resulted in the contamination have not been identified. Furthermore, sampling results indicate multiple contamination sources. The following are considered to be possible contamination events and are discussed in greater detail in Sections 4 and 5:

- Dumping of contaminants into a former trench south of well 37;
- Dumping of contaminants into small ponds or pits located near the North Lagoon and
- Dumping of contaminants in a low-relief area that formerly existed near the present locations of well 37 and former locations for wells 58, and 64.

Thus, it is assumed in the conceptual diagram for the introduction of contaminants to groundwater at the BRGFS that multiple contamination events occurred across a fairly large area (Figure 3.25) (DOE 1994d). Justification for the surface dumping scenario is provided by an evaluation of the data from the multiple completion monitoring wells. The contamination in well 37 is concentrated in the upper zone of the aquifer, an unusual circumstance at the BFC. Well 18, just

downgradient, has a more typical contamination pattern, with most of the contamination in the lower zone. These observations can be explained by applying an understanding of chlorinated-solvent migration behavior to the hydraulic information available from the multiple completion wells. When a large quantity of chlorinated solvents is spilled or released to the ground surface, the solvents will migrate through the soil to the capillary fringe of the water table (Aurelius and Brown 1987). At the capillary fringe, a sufficient mass of solvent must accumulate to overcome the repulsive effects of water before the solvents can break through the boundary of the water surface. As a result, a substantial amount of chlorinated solvents can accumulate at the water-table surface, where they are dissolved and transported by groundwater. With enough accumulated mass, however, the solvents may break through the water-table surface and migrate vertically through the aquifer due to their high densities relative to water.

The field observations and literature review support the conclusion that well 37 was installed in a source area. Once spilled on the surface, chlorinated solvents migrated under the force of gravity to the capillary fringe. In this location, the mass of chlorinated solvent was too small for significant quantities of undissolved liquid (i.e., DNAPL) to migrate through the saturated zone under the force of gravity. When migration as a pure liquid was stopped by the water table, the slightly water soluble solvents were dissolved from the capillary fringe. Once dissolved, solvents no longer sink due to gravity because the density of the contaminated water is so similar to that of clean water (Schwille 1988; Cline and Vista 1985). Therefore, the migration of the contamination from the top of the aquifer at well 37 to the bottom of the aquifer at well 18 is explained by the vertical hydraulic gradient that exists in the area (DOE, 2009). The contaminants are then conducted to the Blue River where they are subject to additional degradation at the groundwater/surface water interface as shown in Figure 3.20. Any remaining contamination would then be overwhelmed by the quantity of dilution from the stream.

3.13.2 ICGFS

The principal contaminants in the ICGFS are dense, chlorinated solvents such as TCE, 1, 2-DCE and vinyl chloride. Traces of benzene, other petroleum hydrocarbons, and PCBs have also been found. Figure 3.26 presents a conceptual diagram of how the contaminants entered and spread in the subsurface from the TCE Still Area, Oil House, and Classified Waste Burial Trenches. This figure serves as a generalized conceptual model for contamination because it includes a range of contaminant entry mechanisms that were present in other solid waste management units in the area. These mechanisms include spilling of waste on the ground surface, disposal and resultant leakage into leaky subsurface drains, and release of waste material either from a leaking tank, abandoned sump, or other burial mechanism (Figure 3.26). Because several wells, notably KC87-69 and KC94-192 show much higher concentrations in the lower zone, it is assumed that, in contrast to the BRGFS, some of these spills were large enough to break through the water-table surface and migrate vertically through the aquifer.

Petroleum hydrocarbons such as benzene, toluene, ethylbenzene and xylene (BTEX) and total petroleum hydrocarbons (TPH) behave differently than do chlorinated solvents due to their low densities relative to water. If spilled in large quantities, they migrate downward until they encounter the water-table surface. Because the densities of most petroleum hydrocarbons are less than water, they float on the water-table surface. Small quantities will dissolve and be advected by the local groundwater flow. These compounds also degrade more rapidly than chlorinated solvents. In the absence of pumping, contaminants will eventually reach Indian Creek where they are subject to additional degradation at the groundwater/surface water interface as with the Blue River. Any remaining contamination would then be overwhelmed by the quantity of dilution from the stream.

3.14 Potential Co-solvent Effects

The potential effect of cosolvency on the mobility of PCBs at the BFC was raised by regulatory reviewers in 1992 (letter to Phil Keary, U.S. DOE, from Kenneth S. Ritchey, U.S.EPA, May 5, 1993) and was addressed in the TCE Still Area RFI Report (DOE 1994c). Because of the previous regulatory concern, the potential effect of co-solvency is summarized in this subsection. There are locations at the BFC (e.g. TCE Still Area) where relatively high concentrations of dissolved chlorinated solvents may come in contact with non-aqueous PCBs that have accumulated at the bottom of the alluvial aquifer. Although PCBs have a higher molecular weight than the chlorinated solvents, the solvation process in water is similar at the molecular level and a relatively non-polar organic compound with a significant amount of chlorine must be dissolved in either case. What occurs can be understood by viewing very simple systems. For example, a solution saturated with TCE (the maximum amount has dissolved) would contain approximately 1000 mg/L of TCE. If this solution comes in contact with a non-aqueous pool of TCE, more TCE cannot dissolve because the solution is already saturated. In a like manner, a saturated solution of TCE will not enhance the solubility of similar compounds such as other chlorinated solvents or even PCBs. If it did, it would have to be capable of enhancing the solubility of itself, which would mean that it would ultimately have to be miscible with water. In other words, rather than a co-solvency effect, TCE and PCBs are competitors for the small capacity of water to dissolve such compounds.

These considerations explain why co-solvency studies have focused on water-miscible cosolvents such as methanol (Means et al. 1989; Rao et al. 1990, Brusseau et al. 1991; Dickhut et al. 1991, Rao et al. 1985, Nkedi-Kizza et al. 1985). Water-miscible cosolvency is not a surrogate for the solvation mechanism that would have to be operative for chlorinated solvents in water to enhance the solubility of PCBs. Nevertheless, the results that have been obtained provide a useful reference point from which to examine the issue of co-solvency. The co-solvency mechanism for water-miscible co-solvents is based on the added solvent

having a significant dipole moment. Methanol, because of the -OH group, has sufficient polarity to interfere with the hydrogen bonds in water. The methanol molecule is sufficiently polar to be soluble with water in all proportions. A mixture of water and methanol, therefore, contains fewer hydrogen bonds (less ionic character) than water alone and because of the -CH₃ group has less polarity (more organic character) than water alone. Consequently, a mixture of water and methanol can enhance the solubility of hydrophobic compounds such as PCBs. However, studies found that the cosolvent must be present in concentrations of several percent or greater. In other words, concentrations of polar, miscible solvents greatly exceeding the solubility of TCE (which is approximately 0.1%) are required to show a significant co-solvency effect. The conclusion from one of these investigations is illustrative of this point: "...co-solvents showed significant reproducible sorption and attenuation reductions but probably are ultimately not important to overall solution fate" (Staples and Geiselman 1988).

3.15 Conceptual Model of Contamination

Similar to the steps employed to create the site groundwater flow model, a groundwater contaminant transport model first required the creation of a conceptual model of contaminant transport. Specifically, a conceptual model is created to describe locations and quantities of TCE contamination that enters the groundwater flow system and the factors controlling contaminant movement. The conceptual model of contaminant transport, derived from review and assimilation of site specific data produces a series of statements that will serve to help others understand contaminant behavior at the BFC.

The conceptual model is as follows:

With regard to TCE source areas:

- TCE source areas are located in both the upper clayey-silt and basal gravel but due to the age of releases a greater quantity of DNAPL is believed to be located in the basal gravel.

- BFC source areas are considered constant. While the volumes of subsurface DNAPL have never been quantified, based on only minor changes in dissolved contaminant concentrations (despite almost two decades of pumping well operation), the DNAPL volumes are sufficient to be considered constant sources. That is the DNAPL is not expected to be depleted anytime in the near future. This consistency is also corroborated in Annual Groundwater Corrective Action reports that show that dissolved contaminant mass does not decrease appreciably over time (DOE, 2012).
- The TCE sources have been active for sufficient time such that the loading rates are constant.
- The former drainage ponds and ditches (i.e., Old Ponds) are the primary TCE source areas for the NEA (see Sections 4 and 5 for more discussion).
- There are numerous (some high concentration) TCE source areas in the industrialized portion of the plant.

With regard to plumes:

- The plumes are mature enough so that they have reached steady-state in that concentrations and distributions within the plumes are relatively constant.
- TCE, 1, 2-DCE and vinyl chloride are the primary volatile organics of concern at the KCP.
- 1, 2-DCE is a degradation product of TCE. Vinyl chloride is a degradation product of 1, 2-DCE.

With regard to contaminant transport properties:

- Based on concentration distributions, TCE and vinyl chloride are expected to have shorter half-lives than DCE.
- The cattail region exhibits much faster biodegradation rates relative to the remainder of the aquifer based on the limited nature of any contamination north of this feature.
- 1,2-DCE migration is more retarded than vinyl chloride which is more retarded than TCE.

3.16 Groundwater Use

This section describes groundwater use near the BFC. It is important to realize that the problem of high arsenic in Midwestern alluvial groundwater had not yet

been recognized when environmental cleanup work at the BFC began. The State of Missouri was gathering data as documented in reports prepared at that time (Korte 1990), but, the extent of the problem was not known and acknowledged nor publicized until the mid-1990's. Now, however, it is unlikely any wells could be approved for human consumption in the alluvium near the BFC because of this problem.

The only groundwater pumping wells near the KCP are the pumping wells that comprise the BFC groundwater pumping system (Plate 1). These wells extract contaminated groundwater from the federal complex and are not used for water supply. Treated groundwater is discharged to the sanitary sewer system.

In 1995 DOE conducted a literature review and contacted ten federal and state agencies and two drilling contractors concerning water usage in and adjacent to Jackson County, Missouri (DOE 1995d), (the drilling companies were unwilling to go through their back records) (DOE 1986b). This search revealed that there were no public water supply wells within 4 miles (6.4 km) of the BFC. It is also unlikely that there are any privately owned, potable-water-supply wells in use nearby. There is some uncertainty about the existence of such wells because they were formerly unregulated and the State of Missouri required no documentation. The Kansas City Department of Environmental Hazards informed BFC personnel that the only method for locating domestic water wells was by door-to-door searching in the area of interest (DOE 1986b). Consequently, a door-to-door survey and a mailing to nearby residents were conducted in the fall of 1990. All of the respondents to this survey were asked if they or their neighbors had a water well. No domestic or industrial uses of the groundwater were documented.

A comprehensive literature review performed by DOE (1986b) early in the site characterization effort also showed no use of groundwater in the Blue River alluvium of Jackson County and only limited use of bedrock aquifers in nearby Johnson County, Kansas (Spruill and Kenny 1981; O'Connor 1971). Domestic

wells screened in bedrock aquifers of Johnson County have low yields of moderately saline water. It is unlikely that wells such as these are in use near the BFC because Kansas City municipal water supplies are readily available.

In 1989, EPA Region VII provided a list of alternate water supplies in the Kansas City area. The list was obtained from the Federal Emergency Preparedness Agency. A reconnaissance of this list of wells was performed. Only two of the wells were within 5 miles (8 km) of the BFC and both were completed in formations that could not be affected by formations present at the BFC. One well is completed slightly above the Knobtown Sandstone in a formation that is absent at the BFC. This well is 3 miles (4.8 km) west and 2 miles (3.2 km) downdip from the plant. The second well is completed approximately 60 ft above the Knobtown Sandstone and is also in a unit that is absent at the BFC. This well is 5 miles (8 km) southwest of the Complex.

Subsequent to the previous work, the GSA performed a Preliminary Assessment and Site Inspection (PA/SI) (SCS 2008) which re-visited groundwater use in the vicinity of the BFC. That report corroborated the previous information. The MDNR and the Kansas Department of Health and Environment (KDHE) were contacted to determine the locations of water wells within four miles of the KCP. Eleven wells were identified; two of these being domestic wells:

- Domestic well located approximately 8,600 feet to the north in Missouri (depth 100 feet).
- Domestic well located approximately 18,000 feet to the northeast in Missouri (depth 310 feet).

Considering both the distance from the KCP and the depth; these wells could not be affected by shallow alluvial groundwater from the BFC. The PA/SI report concluded the groundwater usage section with the following: “There are no public drinking water wells or wellhead protection areas located [within a four-mile

radius of the BFC]. The majority of residents in Jackson County, Missouri, and Wyandotte and Johnson County, Kansas obtain their drinking water from municipal providers. Water District Number 1 of Johnson County obtains its drinking water from surface water intakes and wells along the Kansas and Missouri rivers, which are approximately 13 miles and 17 miles northwest of the [BFC], respectively. The Board of Public Utilities (BPU, Kansas City, and Kansas) and Kansas City, Missouri, obtain their drinking water from surface water intakes and wells along the Missouri River. The City of Independence, Missouri, obtains its water from alluvial wells along the Missouri River. The Kansas City, Missouri, wells and intake are approximately 12 miles north of the [BFC]; the BPU intakes and wells are more than 14 miles northwest of the [BFC]; and the Independence wells are more than 15 miles northeast of the [BFC]. With the exception of the Independence wells, all the public water supply sources are upstream of the confluence of the Blue and Missouri Rivers. The Independence wells are more than 16 miles downstream of the [BFC]. All other public water sources on the Missouri River are further downstream than the City of Independence, Missouri, wells.

Based on the above information:

- No wells are located within a 1-mile radius; the site itself is on municipal water.
- One domestic well, serving up to 8 people, and two heat pump wells are located within a 1 to 2-mile radius.
- One irrigation well and one heat pump well are located within a 2 to 3-mile radius.
- One domestic well, serving up to 8 people, and five heat pump wells are located within a 3 to 4-mile radius.

In summary, all indications from the above agencies and contractors and from the literature were that there were no potential impacts to domestic groundwater supplies from the BFC. Because of the low yield and moderately saline nature of the groundwater; there is little likelihood these circumstances would change.

3.17 Current Surface Water

Note: A complete discussion of surface water conditions is provided under Section 6.0

The BFC is bounded on the east by the Blue River and on the south by Indian Creek, which joins the Blue River just southeast of the BFC. Immediately downstream from this confluence, a U. S. Geological Survey (USGS) stream gauging station has been in operation since 1939. Data from this station show that low flows are supplemented by wastewater discharges.

The Blue River is not a known source for residential or commercial water. The MDHSS currently maintains a fish consumption advisory applicable to segments of Indian Creek and the Blue River adjacent to the BFC (see Section 8.6).

A downstream groundwater and surface-water industrial usage survey was completed in 1991 (DOE 1995d). Results from the survey indicate that all of the businesses contacted obtained their water from the city and intend to continue doing so. Additionally, none of these businesses use the Blue River for recreational or production purposes (DOE 1992a).

Anecdotal evidence based on many years of monitoring the river indicate only sporadic use by fisherman and no use for recreational boating. Similarly, a door-to-door survey performed in 2003 showed little recreational use of Indian Creek by nearby residents though some people do fish on the Blue River and children may, at times play, in Indian Creek (MDNR 2006). Finally, the recent PA/SI conducted for the GSA (SCS 2008) also examined surface water use and came to the same conclusions.