Remedial Investigation of the Vienna Wells Site:

Maries County, Missouri, 2011–2016

Administrative report prepared for the U.S. Environmental Protection Agency

Final Draft, June 2017

Contract No. DW1495797201-5

VOLUME I

Prepared by:

Jordan Wilson, Hydrologist, U.S. Geological Survey, Missouri Water Science Center

USGS

science for a changing world

Cover photographs taken by Jordan Wilson of the U.S. Geological Survey, Rolla, Missouri.

1. Slotted-floor drains at the southwest corner of the former hat factory (2012).

2. Drum inside basement of the former hat factory building (2013).

3. Tree-coring sampling near the former hat factory (2013).

Contents

Figures

Tables

Abbreviations

AB	air blank
ALS	above land surface
bgs	below ground surface
BH	borehole
BOH	bottom of hole
BOW	bottom of well
BRA	Baseline Risk Assessment
CAS	Chemical Abstracts Service
CFC-113	1,1,2-Trichloro-1,2,2-trifluoroethane
cis-DCE	*cis*-1,2-dichloroethene
COC	contaminant of concern
COPC	chemicals of potential concern
CS	cistern
CU	cuttings
DEM	digital elevation model
DL	detection limit
DR	drill water
DW	domestic well
E	estimated
ECD	electron capture detection
EPC	exposure point concentration
ERA	Ecological Risk Assessment
ERC	Environmental Research Center
ESV	ecological screening value
EX	excavation
FS	Feasibility Study
FSP	Field Sampling Plan
GC	gas chromatograph
GW	groundwater
HHRA	Human-Health Risk Assessment
HQ	hazard quotient
HSL	Hazardous Substance List
K	hydraulic conductivity
LMW PAH	low molecular-weight, polycyclic-aromatic hydrocarbon
LOWESS	locally weighted scatterplot smoothing
M	constituent presence verified but not quantified
MCL	maximum contaminant level
MCLG	maximum contaminant level goal
MDL	method detection limit
MDNR	Missouri Department of Natural Resources

MRL	method reporting level
MS&T	Missouri University of Science and Technology
MO WSC	Missouri Water Science Center
MW	monitoring well
NA	not applicable
NAD 83	North American Datum of 1983
NAVD 88	North American Vertical Datum of 1988
ND	not detected
NOAA	National Oceanic and Atmospheric Administration
NWIS	National Water Information System
NPL	National Priorities List
PAH	polycyclic-aromatic hydrocarbon
PCE	tetrachloroethene
PDMS	polydimethylsiloxane
PL	pool
PS	public-supply
PVC	polyvinyl chloride
QA	quality assurance
QAPP	Quality Assurance Project Plan
QC	quality control
ρ	rho, Spearman correlation coefficient
R	presence or absence of constituent could not be determined because of severe quality-control issues
R^2	Pearson's coefficient of determination
RESV	refined ecological screening value
RI	Remedial Investigation
RPD	relative percent difference
S	shrub
SCH	schedule
SE	seep
SI	Site Investigation
SN	sanitary sewer
SO	soil
SP	spring
SPME	solid-phase microextraction
SS	split-spoon
ST	stream
SUB	subsurface
SUR	surficial
SVOC	semivolatile organic compounds
SW	surface water
T	tree
TCE	trichloroethene

trans-DCE	*trans*-1,2-dichloroethene
TP	tap
TR	trench
TVOC	total volatile organic compounds
U	not detected above the detection limit
UNK	unknown
USDA	U.S. Department of Agriculture
USEPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey
WA	water
VOC	volatile organic compound

Unit Abbreviations and Symbols

--	no data available
%	percentage
<	less than
>	greater than
°C	degree Celsius
°F	degree Fahrenheit
ft	foot
ft/d	foot per day
ft/ft	foot per foot
ft/mi	foot per mile
ft/yr	foot per year
ft^2	square foot
ft^3	cubic foot
ft^3/s	cubic foot per second
g	gram
gpm	gallon per minute
in.	inch
kg	kilogram
km	kilometer
lb	pound
µg/kg	microgram per kilogram
µg/L	microgram per liter
$µg/m^3$	microgram per cubic meter
µL	microliter
µS/cm	microsiemens per centimeter at 25 degrees Celsius

m	meter
mi	mile
mi^2	square mile
mg/kg	milligram per kilogram
mg/L	milligram per liter
mL	milliliter
ng/L	nanogram per liter
oz	ounce

Conversion Factors

Multiply	By	To obtain
Length		
inch (in.)	2.54	centimeter (cm)
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
Area		
acre	0.4047	hectare (ha)
acre	0.4047	square hectometer (hm^2)
acre	0.004047	square kilometer (km^2)
square foot (ft^2)	0.09290	square meter (m^2)
square yard (yd^2)	0.836127	square meter (m^2)
square mile (mi^2)	2.590	square kilometer (km^2)
Volume		
ounce, fluid (fl. oz)	0.02957	liter (L)
gallon (gal)	3.785	liter (L)
gallon (gal)	0.003785	cubic meter (m^3)
gallon (gal)	3.785	cubic decimeter (dm^3)
cubic foot (ft^3)	28.32	cubic decimeter (dm^3)
cubic foot (ft^3)	0.02832	cubic meter (m^3)
cubic yard (yd^3)	0.7646	cubic meter (m^3)
Flow rate		
foot per second (ft/s)	0.3048	meter per second (m/s)
foot per day (ft/d)	0.3048	meter per day (m/d)
foot per year (ft/yr)	0.3048	meter per year (m/yr)
foot per mile (ft/mi)	0.18939	meter per kilometer (m/km)
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)
cubic foot per day (ft^3/d)	0.02832	cubic meter per day (m^3/d)
gallon per minute (gpm)	0.06309	liter per second (L/s)
gallon per day (gal/d)	0.003785	cubic meter per day (m^3/d)
Mass		
pound, avoirdupois (lb)	0.4536	kilogram (kg)
Hydraulic conductivity		

Multiply	By	To obtain
foot per day (ft/d)	0.3048	meter per day (m/d)

International System of Units to U.S. customary units

Multiply	By	To obtain
Length		
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
meter (m)	1.094	yard (yd)
Volume		
microliter (μL)		ounce, fluid (fl. oz)
milliliter (mL)		ounce, fluid (fl. oz)
Mass		
gram (g)	0.03527	ounce, avoirdupois (oz)
kilogram (kg)	2.205	pound avoirdupois (lb)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

°F=(1.8×°C)+32

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:

°C=(°F-32)/1.8

Vertical coordinate information is referenced to North American Vertical Datum of 1988 (NAVD 88).

Horizontal coordinate information is referenced North American Datum of 1983 (NAD 83).

Elevation, as used in this report, refers to distance above the vertical datum.

Specific conductance is given in microsiemens per centimeter at 25 degrees Celsius (μS/cm at 25 °C).

Concentrations of chemical constituents in water are given either in milligrams per liter (mg/L) or micrograms per liter (μg/L).

Concentrations of chemical constituents in soil are given either in milligrams per kilogram (mg/kg) or micrograms per kilogram (μg/kg).

Concentrations of chemical constituents in air are given in micrograms per cubic meter (μg/m^3).

Remedial Investigation of the Vienna Wells Site: Maries County, Missouri, 2011–2016

By Jordan L. Wilson

Executive Summary

Vienna, Missouri, is in central Missouri and lies across a topographic divide between the Gasconade River about 2.0 miles to the east and the Maries River about 1.9 miles to the west-northwest. The former Langenberg Hat Company and Top This, Incorporated property (hereinafter referred to as the "former hat factory" or "former hat factory property") is a subsection of the Vienna Wells Superfund site and is in the northeast part of Vienna near the intersection of 10th Street and Chestnut Street.

The city of Vienna has three public-supply wells: wells COV-01, COV-02, and COV-03 (drilled in 1954, 1985, and 1994, respectively). In 2006, routine water-quality monitoring of the three public-supply wells in Vienna indicated that tetrachloroethene (PCE) had exceeded the U.S. Environmental Protection Agency (USEPA) maximum contaminant level (MCL) of 5.0 micrograms per liter (µg/L) in a sample from well COV-03. After the detection above the MCL, a Site Investigation was completed by the Missouri Department of Natural Resources, who detected PCE in wells COV-01 and COV-03 and concluded that the likely source of PCE contamination was from the former hat factory because PCE potentially was used in factory operations and the factory is near well COV-03. Because of aesthetic concerns involving large iron levels, public-supply well COV-01 was put on standby in the fall of 2007. The pump was removed from well COV-02, and in 2008 the well was converted to a groundwater-level observation well by the Missouri Department of Natural Resources. In October 2012, an air-stripper system was installed at well COV-03 to remove PCE from raw water pumped from wells COV-03 and

COV-01. Although connected to the air-stripper system, well COV-01 was placed on standby and COV-03 is used as the primary water supply.

Three small dumps are in the wooded area on the former hat factory property: (1) the central dump, which is northeast of the former hat factory and characterized by large metal debris, roofing material, old drums, bricks, and various metals cans; (2) the eastern dump, which is on the east side of the former hat factory property and mainly consists of old fur and occasional leather strips; and (3) the metal dump, which is north of the former hat factory and characterized by buried and partially buried metal debris. Material in the central and eastern dumps seems to have been deposited on the surface, whereas material in the metal dump seems to have been deposited in an excavation.

During 2010, the USEPA Region 7 Superfund Branch requested technical assistance from the U.S. Geological Survey (USGS) Missouri Water Science Center to complete a Remedial Investigation (RI) of the Vienna Wells site. This RI report focuses on the former hat factory property, which occupies 7.9 acres in the northeast part of Vienna, Missouri. The purposes of this RI report are to document the extent and magnitude of PCE contamination at the Vienna Wells site, determine the relation between the former hat factory and the contaminated public-supply well (well COV-03), and characterize the human-health and ecological risks posed by contamination at the Vienna Wells site.

The former hat factory was operational from 1952 to 1996. During that time, the facility was owned by two separate companies and at least two solvents (PCE and 1,1,2-trichloro-trifluoroethane) were apparently used sometime between 1952 and 1996 with probable usage from 1965 to 1969. The main aquifer in the Vienna area is the Ozark aquifer, which supplies much of the water to towns, cities, and rural water districts in southern Missouri. In the Vienna area, the Ozark aquifer provides all domestic, industrial, and public water used and is about 1,000 feet (ft) thick. Groundwater in the Ozark aquifer is unconfined in the Vienna area, and groundwater flow directions are strongly affected by regional topography. Regional groundwater flow in the Vienna area is to the northeast toward the Gasconade River. Superimposed on the regional flow system is a shallower flow system controlled by the surface topography in the Vienna area.

Field activities at or near the former hat factory were completed from 2011 to 2016 and included the

installation and sampling of 17 monitoring wells ranging from less than 15 to more than 320 ft deep; collection and analysis of more than 120 samples of drill cuttings from discrete depths during monitoring well installation; more than 400 water samples from completed monitoring, domestic, industrial, or public-supply wells; 488 tree-core samples; 143 surface-water and spring samples; more than 1,150 soil samples from 265 soil borings; 56 wastewater samples from the city sanitary-sewer system; and surveying of 2 nearby wells using downhole geophysical methods. In addition, a reconnaissance sampling of the city water system included the collection of six samples from taps to determine if passage through the distribution system affected the concentration of PCE. More than 145 vapor-intrusion samples were also collected by the USEPA to characterize any potential vapor intrusion into homes in the vicinity of the former hat factory property.

Of the 488 tree-core samples analyzed for PCE, trichloroethene (TCE), *cis*-1,2-dichloroethene, and 1,1,2-trichloro-1,2,2-trifluoroethane, 324 and 56 had detections greater than the method detection limit for PCE and TCE, respectively. Trees with large PCE concentrations (greater than 1,000 nanograms per liter) were clustered near the former hat factory to the north and south, near the central dump, and near the eastern property line. The largest PCE concentrations were measured in a sycamore tree near the central dump and in three trees directly south of the slotted floors. Of the 1,185 soil samples collected by the USGS and USEPA for laboratory and portable gas chromatograph analysis, PCE was detected in 25 surficial soil samples and 284 subsurface soil samples. At and in the immediate vicinity of the former hat factory, concentrations of PCE were largest below the concrete slab near the southwest corner of the former hat factory, north and northwest of the three trees that contained the largest PCE concentrations in tree-core samples. Of the 93 soil samples collected from boreholes beneath the slotted floor and beneath the solid slab south of the slotted floor, all had PCE concentrations that ranged from less than detection to 51,840 micrograms per kilogram (µg/kg) in portable gas chromatograph samples collected from 0.6 to 12.3 ft below ground surface (bgs).

Northeast of the former hat factory near the central dump, PCE concentrations in soil samples ranged from less than detection to 15,552 µg/kg in portable gas chromatograph samples, and sample depths ranged from 0 to 14.6 ft. Most PCE detections in soil samples were between 4 and 11.5 ft, and the largest PCE concentrations in soil samples were between 10 and 11.2 ft. Concentrations of PCE generally decreased with depth in this area

and were all below detection until 4 ft bgs, indicating contamination likely came from local dumping at depth.

A general absence of PCE in soil samples from the east part of the former hat factory property conflicts with the tree-core data, which indicates areas of probable subsurface PCE contamination in the east part of the site extending off the property to the east. Because PCE concentrations in drill-cutting samples collected during the construction of well MW-05 increased from 5.0 µg/kg between 5 and 10 ft bgs to around 200 µg/kg at 65 ft bgs, it is likely that the PCE concentrations in tree-core samples are a result of PCE vapors and groundwater transport into trees below the restrictive sandstone layer from the same source as the plume from the former hat factory.

Several contaminants of concern in surficial soil samples were identified in the risk assessment including the polycyclic-aromatic hydrocarbon (PAH) compounds benzo[*a*]anthracene, benzo[*a*]pyrene, benzo[b]fluoranthene, dibenzo[*a,h*]anthracene, and Ideno[1,2,3-*cd*]pyrene; and the metals aluminum, arsenic, chromium, manganese, and thallium. Most PAH and metal detections were in the vicinity of the former hat factory and near the metal and central dumps north and northeast of the former hat factory—except a large chromium concentration in a sample from the eastern dump on the east side of the former hat factory property that is presumably related to the piles of old fur and leather strips, which are known to contain chromium when tanned. The large concentrations of PAHs and metals in the metal and central dumps corroborates field observations of roofing material, large metal debris, old drums, bricks, and various metal cans in these dumps.

An estimated 1,500,000 cubic feet or 56,000 cubic yards of PCE-contaminated soils are above the USEPA protection-to-groundwater threshold of 5.1 µg/kg, and about 700,000 cubic feet or 26,000 cubic yards of soils have PCE concentrations greater than 100 µg/kg. These soils contain a total estimated 33 kilograms (kg) (about 5.5 gallons of pure product) of PCE.

Although PCE was detected in several soil samples adjacent to the 1 ft deep, collapsed sewer line near the central dump, concentrations of PCE generally increased with depth and were all below detection until 4 ft bgs; the five samples that were above 100 µg/kg were at or below 6 ft bgs, suggesting contamination may have originated from local dumping at depth. Although the largest PCE concentrations in the central dump are more than 30 ft lower in elevation from the largest PCE concentrations under the slab of the former hat factory, it is

unlikely that PCE migrated through the soil from under the slab because of the large distance and similarity in magnitude of PCE concentrations. Additionally, all the boreholes between the south part of the former hat factory and the central dump did not contain large PCE concentrations, although land surface slopes heavily downward toward the central dump. It is possible, however unlikely, that PCE could have migrated from the slab to the central dump through preferential pathways. Another potential origin of PCE may be from PCE disposal in the sewer line. Additional soil sampling in and in the vicinity of the central dump, and between the central dump and the former hat factory to elevations comparable to samples collected from and in the vicinity of the central dump, may be necessary to determine the source of PCE contamination in the central dump.

Although volatilization and some reductive dechlorination of PCE in contaminated soils has occurred, these processes seem to be slow; after potentially 48 to 65 years of residence time in soils, PCE still composes almost 90 percent of the total chlorinated volatile organic compounds in the vicinity of the former hat factory. Because substantial PCE concentrations remain in soils under the concrete slab of the former hat factory, mechanisms for PCE loss from the soil will not remove a substantial mass of the remaining PCE from the soils for decades to come, and these soils will be a continued source of PCE to groundwater.

Results of the RI of the Vienna Wells site indicate the presence of substantial contamination in soils and groundwater that present unacceptable human-health risks and hazards as determined by the Baseline Risk Assessment completed by the Black and Veatch Special Projects Corporation. The primary source area is a PCE source area in soils under the slotted-floor section of the former hat factory. Within this area, PCE was detected in two soil samples above the USEPA Region 9 residential screening level for soil of 24,000 µg/kg. Concentrations of PCE in soils generally increased with increasing depth; however, concentrations of PCE decreased with depth in the vicinity of the slotted-floor part of the former hat factory where they were the largest measured at the site. The origin of PCE at the Vienna Wells site was the apparent use of PCE as a solvent in the hat-making process. Because only two samples exceeded the residential screening level for soil, PCE in soil was determined to not present unacceptable cancer risk or noncancer hazard sitewide to future residents and current and future workers; however, concentrations of several PAHs and metals (arsenic and chromium) pose an excessive cancer risk in soil, and manganese poses an excessive noncancer risk in soil.

Analyses of groundwater samples collected from monitoring wells at the Vienna Wells site indicate that a plume of PCE contamination in shallow (greater than about 710 ft elevation) groundwater extends from the former hat factory to the east to a small creek where it likely discharges in seeps and springs, including spring WP-020. Concentrations of PCE in wells MW-01 and MK-01 suggest some migration to the south. Within the plume, PCE concentrations in the bedrock aquifer ranged from about 1,000 µg/L directly below and in the immediate vicinity of the former hat factory to less than 100 µg/L about 200 ft south of the former hat factory and decreased to less than 10 µg/L about 125 ft north of the former hat factory. The plume extends deeper into the aquifer affecting nearby public-supply wells COV-01 and COV-03 that are cased more than 300 ft bgs. Presumably, pumping from these public-supply wells captures most PCE migrating deeper into the aquifer. Pumping from well COV-01 likely resulted in drawdown of the water table before the well was put on standby in 2007. Because there is no gap in pumping from public-supply wells COV-01 (1954-2007) and COV-03 (1994-present [2017]), and because of the drawdown (about 0.5 ft) in monitoring wells installed beneath the hat factory building slab (MW-06A and MW-06) during pumping of COV-03, most contaminated groundwater at depth (less than about 710 ft elevation) at the Vienna Wells site likely always has been captured by one of these two public-supply wells. More deep monitoring wells would need to be drilled between the monitoring wells on the hat factory property and public-supply wells to verify this spatial distribution of PCE in deeper groundwater. The extent of the plume in the shallow subsurface (less than about 710 ft elevation) is more affected by topography and may not be captured by the pumping of public-supply wells COV-01 and COV-03. Based on the completed exposure pathways and calculations made in the human-health risk assessment, groundwater at the Vienna Wells site presents the largest unacceptable cancer risk of 4×10^{-4} and noncancer hazard quotient of 17 to future residents and workers because of potential ingestion and dermal contact with benzene, dieldrin, PCE, TCE, and chromium through groundwater.

The area of the groundwater plume containing PCE concentrations above the 5 µg/L MCL is about 12 acres in shallow groundwater and contains an estimated 4.5 kg or 9.9 pounds of PCE. The mass of PCE in deeper groundwater could not be estimated because of a lack of data; however, PCE concentrations and monthly pumping data obtained from public-supply wells COV-01 and COV-03 were used to estimate the total mass of

PCE removed from groundwater in the 20 years between July 1995 (first groundwater sample with PCE below the MCL in public-supply well COV-01) to June 2015 to be about 7.1 kg or 16 pounds. No significant trends in PCE concentration with time were observed at any groundwater monitoring wells, springs, or seeps from 2012 to 2016, likely because of the short period of data collection; however, PCE concentrations in public-supply well COV-03 before treatment had a significant upward trend from 1997 (first groundwater sample with PCE below the MCL in public-supply well COV-03) to 2016.

Based on the generally aerobic conditions in most of the bedrock aquifer and the predominance of PCE as compared to its degradation products, TCE and *cis*-1,2-dichloroethene, in groundwater samples, microbial degradation of PCE is not substantial in the bedrock aquifer.

Groundwater flow in the vicinity of the former hat factory property source area is mostly controlled by the pumping of well COV-03, especially in deeper groundwater (less than about 710 ft in elevation) and, to a lesser extent, topography. In contrast, groundwater flow to the east is mainly controlled by topography. Vertical hydraulic gradients estimated from water levels in clustered wells are generally large and indicate that vertical hydraulic conductivity is much smaller than horizontal hydraulic conductivity; however, some areas, such as near well cluster MW-03, may have small vertical hydraulic gradients, indicating increased potential for vertical transport. The average vertical groundwater velocity in the bedrock aquifer at the Vienna Wells site is about 7.4 feet per year, and it would take about 53 and 44 years for groundwater to migrate from the source area underneath the former hat factory to public-supply wells COV-01 and COV-03, respectively. The earliest probable PCE disposal at the former hat factory is 1965, about 29 years before the first sample with a PCE concentration of 0.90 µg/L (at reporting limit) in public-supply well COV-01 and 32 years before the first sample with a PCE concentration of 0.50 µg/L (at reporting limit) in public-supply well COV-03. Because pumping of the public-supply wells may substantially decrease the total transport time, the general agreement in the estimated transport time from the former hat factory property and the initial occurrence of PCE in the two public-supply wells supports the conclusion that contamination from the former hat factory property is the cause of increased PCE concentrations in both public-supply wells. The large mass of PCE in soil (33 kg) compared to groundwater (7.1 kg pumped from 1995 to 2015 and 4.5 kg currently [2017] in shallow groundwater) coupled with a lack of

degradation indicates that contamination will likely be persistent for many decades.

Current (2017) PCE concentrations in the water distribution system with the implementation of the air stripper are below the MCL for PCE. Based on historic PCE concentrations before treatment, if use of the air stripper were to be discontinued then PCE concentrations in finished water entering the distribution system would likely increase to PCE concentrations before treatment, which are above the MCL.

Overall, concentrations of PCE in vapor-intrusion samples were low, and all field blank results were below reporting limits. Tetrachloroethene was detected in several indoor-air and subslab or crawlspace samples at concentrations ranging from 0.46 to 12.0 micrograms per cubic meter ($\mu g/m^3$); however, only one indoor-air sample collected from site VIN-VI-16 contained PCE (12.0 $\mu g/m^3$) larger than a total cancer risk level of 1×10^{-6} (11.0 $\mu g/m^3$). Small concentrations of PCE in vapor-intrusion samples suggest little off-site migration of PCE to the north, west, and south. Although degradation of PCE to TCE was determined to be small, some degradation is occurring, and because the 1×10^{-6} risk of cancer occurs at a small TCE concentrations (0.48 $\mu g/m^3$) compared to PCE (11 $\mu g/m^3$), two homes adjacent to the former hat factory property had concentrations above the 1×10^{-6} risk of cancer because of the occurrence of TCE. Several off-site residential homes had hazard quotients exceeding 1 and ranged from 0.2 to 7 because of vapor intrusion of various volatile organic compounds other than PCE. One location, directly southeast of the former hat factory property, has excessive risk of cancer because of concentrations of 1,2-dichloroethane, benzene, bromodichloromethane, carbon tetrachloride, and chloroform; however, the occurrence of these contaminants at large concentrations is likely the result of contamination from indoor sources. Multiple lines of evidence described in the USEPA vapor-intrusion summary report were evaluated for samples that exceeded the USEPA Region 7 action level of 2.0 $\mu g/m^3$ for TCE to determine whether or not the vapor-intrusion pathway was complete. Based on the vapor-intrusion data and multiple lines of evidence presented, the USEPA considered the vapor-intrusion assessment complete and recommended no further action.

1.0 Introduction

Vienna, Missouri, has a population of 610 (U.S. Census Bureau, 2010a) and is in central Missouri (fig. 1-01). The town surrounds the intersection of U.S. Highway 63 and Missouri State Highway 42 and is at an elevation of about 860 feet (ft) referenced to the North American Vertical Datum of 1988 (NAVD 88; Bachle, 2009). Vienna is the county seat in Maries County and has a total area of 1.06 square miles (U.S. Census Bureau, 2010b). The town lies across a topographic divide between the Gasconade River about 2.0 miles (mi) to the east and the Maries River about 1.9 mi to the west-northwest (fig. 1-01). The former Langenberg Hat Company and the Top This, Inc. building and property (hereinafter referred to as the "former hat factory" and "former hat factory property", respectively) is a subsection of the Vienna Wells Superfund site (hereinafter referred to as the "Vienna Wells site") and is in the northeast part of Vienna near the intersection of 10th Street and Chestnut Street (fig. 1-02).

Figure 1-01. Location of the city of Vienna, Maries County, and a National Oceanic and Atmospheric Administration (NOAA) weather station, Missouri.

Three public-supply wells are operated by the city of Vienna, wells COV-01, COV-02, and COV-03 (fig. 1-02), drilled in 1954, 1985, and 1994, respectively. In 2006, routine water-quality monitoring of public-supply wells in Vienna indicated that tetrachloroethene (PCE) had exceeded the U.S. Environmental Protection Agency (USEPA) maximum contaminant level (MCL) of 5.0 micrograms per liter (µg/L) in one sample from well COV-03 (fig. 1-03). After the detection, a Site Investigation (SI) was completed by the Missouri Department of Natural Resources (MDNR), which detected PCE in city of Vienna public-supply wells COV-01 and COV-03. The SI concluded that the likely source of PCE contamination was the former hat factory because PCE potentially was used in factory operations and the factory is near well COV-03 (Jackson, 2009). Because of aesthetic concerns involving large iron levels, public-supply well COV-01 was put on standby in the fall of 2007 (fig. 1-03). The pump was removed from well COV-02 and replaced with a float and encoder in 2008 to monitor water levels (Jackson, 2009). In October 2012, an air stripper system was installed to remove PCE from raw water at well

COV-03, the only remaining source of drinking water for the city. Although connected to the air-stripper system, well COV-01 was placed on standby and COV-03 is used as the primary water supply.

Figure 1-02. Local points of interest and the former hat factory property, Vienna, Missouri.

Figure 1-03. Concentration of tetrachloroethene (PCE) in Vienna public-supply wells COV-01 and COV-03 before treatment, January 1994 through January 2016.

During 2010, the USEPA Region 7 Superfund Branch requested technical assistance from the U.S. Geological Survey (USGS) Missouri Water Science Center (MO WSC) to complete a Remedial Investigation (RI) of the Vienna Wells site. The USGS prepared an approved scope of work for the RI (USEPA, 2011a), completed field investigations, and prepared this site RI report. The Feasibility Study (FS) will be prepared by a USEPA contractor, and the USGS will provide technical assistance, as needed, through the FS, Proposed Plan, and Record of Decision documents.

1.1 Purpose and Scope of the Vienna Wells Remedial Investigation

This RI report focuses on the former hat factory property at the Vienna Wells site, including 7.9 acres in the northeast part of the city of Vienna, Missouri (fig. 1-02). This RI report documents the extent and magnitude of PCE contamination at the Vienna Wells site, identifies the relation between the former hat property and the contaminated public-supply well COV-03, and characterizes the human-health and ecological risks posed by contamination at the Vienna Wells site.

Field activities at or near the Vienna Wells site were completed from 2011 to 2016 and included the installation of and sampling of groundwater from 17 monitoring wells with depths ranging from less than 15 ft to more than 320 ft; collection and analysis of more than 120 drill-cutting samples from discrete depths during monitoring well installation; collection of more than 400 water samples from completed monitoring, domestic, industrial, or public-supply wells; collection of 488 tree-core samples; collection of 143 surface-water and spring samples; collection of more than 1,150 soil samples from 265 soil borings; collection of 56 samples from the city

sanitary-sewer system; and geophysical logging of 2 monitoring wells. In addition, six water samples were collected at residential taps within the city limits to determine if passage through the water distribution system affected the level of PCE detected in drinking water. More than 145 vapor-intrusion samples also were collected by the USEPA to characterize any potential vapor intrusion into homes in the vicinity of the former hat factory property. Although much of the laboratory data is presented in tables in appendixes included with this report, data from all surface-water and groundwater sites not available in tables in this report are available in the USGS National Water Information System (http://dx.doi.org/10.5066/F7P55KJN), as well as online at *https://mo.water.usgs.gov/*. Site aliases used in this report and their corresponding names in the National Water Information System are in appendix D, table D-01. A summary of previous investigations is presented in the body of this RI report and, where applicable and relevant, data from these investigations are included.

This RI is a cooperative effort between the USGS, USEPA, USEPA contractors, and MDNR. The general geohydrology and extent and magnitude of PCE contamination were investigated by the USGS. Black and Veatch Special Projects Corporation, a USEPA contractor, completed the Baseline Risk Assessment (BRA), which included Ecological Risk Assessment (ERA) and the Human-Health Risk Assessment (HHRA) components. This RI report was prepared by the USGS with the exception of section 5, which was taken from the BRA prepared by Black and Veatch Special Projects Corporation (Black and Veatch Special Projects Corp., 2016).

1.2 Report Organization

The text of the report is divided into the following seven sections:

- **Section 1.0** describes the Vienna Wells site and the purpose and scope of this RI report; summarizes previous investigations of PCE contamination in the Vienna area; and presents a general description of the physical setting, hydrogeology, and groundwater flow in the Vienna area.

  - **Section 2.0** describes RI activities (including tree-core reconnaissance; stream, spring, and seep sampling; groundwater investigations, including geophysical logging of two wells; soil investigations; sanitary sewer and septic investigations; vapor-intrusion sampling; and the analytical methods used in the RI.

- **Section 3.0** presents detailed information on the nature and extent of contamination at the Vienna Wells site.

- **Section 4.0** describes the fate and transport of contamination.

- **Section 5.0** presents a summary prepared by Black and Veatch Special Projects Corporation of the human-health and ecological risks posed by the Vienna Wells site.

- **Section 6.0** contains the summary and conclusions.

- **Section 7.0** lists the references cited in this RI report.

1.3 Previous Investigations of Contamination at the Vienna Wells Site

The presence, nature, and extent of contamination were investigated from 1991 to 2009 in the Vienna area by a variety of entities. These investigations form the foundation for the RI described in this report.

1.3.1 Discovery of Contamination

The Public Drinking Water Branch of MDNR began routine monitoring of all public-supply wells for volatile organic compounds (VOCs) in the early 1990s. Initially, PCE concentrations in public-supply well COV-01 were below the detection limit on May 20, 1991, and March 22, 1993; increased to 0.9 µg/L on June 7, 1994; and increased further to 3.82 µg/L on February 7, 2006 (fig. 1-03). Well COV-03 was drilled in 1994, and a sample collected after drilling had no PCE detections; however, a sample collected in March 1997 had a PCE concentration of 0.5 µg/L (fig. 1-03). In the 38 samples between September 12, 1994, and August 31, 2006, PCE concentrations in well COV-03 continued to increase, exceeding the MCL for the first time on August 31, 2006. On March 28, 2007, an abbreviated preliminary assessment was completed by the MDNR because of the documented presence of PCE in public-supply wells COV-01 and COV-03 (USEPA, 2010). Because of the proximity of the former hat factory property to public-supply well COV-03, the former hat factory was suspected as a source of the PCE contamination (Jackson, 2009). The Vienna Wells site was proposed for inclusion onto the Superfund National Priorities List (NPL) on November 29, 2007 (USEPA, 2010).

1.3.2 Site Investigation (2007-2009)

On April 1, 2007, the MDNR began an SI to identify potential sources of PCE, determine if the release constituted a substantial risk to human health or the environment, and evaluate the site using the Hazardous Ranking System to determine if proposal to the NPL was appropriate. A series of sampling events during the SI confirmed the presence of PCE in soils at the former hat factory property (Jackson, 2009).

During April 2007, the MDNR completed soil sampling at two locations (borings 01 and 07), and soil-gas sampling at nine locations (borings 01 through 09; fig. 1-04) along the north and west sides of the former hat factory and at a background location from the Vienna Ball Park. During 2008, the MDNR completed soil sampling on April 1 at nine locations (borings 10 through 18) and on August 26 and 27 at eight locations (borings 19 through 26). Soil borings were drilled to refusal (typically between 6 and 12 ft), and a gas probe was used as a screening tool to analyze the presence of VOCs in soil-gas; however, no VOCs were detected in any of the soil-gas samples. Soil samples were collected at refusal. PCE was detected in the soil sample from boring 01, north of the former hat factory, from 10 to 11 ft at 0.198 milligrams per kilogram (mg/kg; fig. 1-05) during the initial data collection.

Figure 1-04. Location of Missouri Department of Natural Resources soil samples collected from April 12, 2007, to August, 26, 2008, at the Vienna Wells site in Vienna, Missouri. Modified from Jackson, 2009.

Figure 1-05. Result summaries of select Missouri Department of Natural Resources soil samples collected from April 12, 2007, to August 26, 2008, at the former hat factory property in Vienna, Missouri. Modified from Jackson, 2009.

Two additional soil sampling efforts were completed at the site by the MDNR on April 1, 2008, and August 26, 2008. In April 2008, soil borings 10 through 18 were advanced north of the former hat factory (fig. 1-04). No screening tools were used, and samples were collected from each boring at refusal. Soil samples from borings 11, 16, and 17 had PCE concentrations ranging from 0.00323 to 16.3 mg/kg. Boring 16, which contained the largest concentration of PCE, was northeast of the former hat factory in a location described by the water

system operator of Vienna as the location of a former wastewater lagoon (appendix A, figs. A-01 and A-02). In August 2008, soil borings 19 through 26 were advanced in the vicinity of boring 16, and eight soil samples were collected at refusal (figs. 1-04 and 1-05). A portable photoionization detector was used to screen samples for total VOCs to determine sample collection depths. PCE was detected in four of the eight samples ranging from 0.00574 to 65.4 mg/kg (Jackson, 2009). Two surface soil samples were also collected and contained PCE concentrations of 0.00719 and 0.0141 mg/kg, which were substantially above background but below health-based benchmarks.

Overall, a total of 22 soil samples were collected during the 3 soil sampling efforts; of those samples, 2 were collected at the surface and 20 were collected at refusal ranging from 6.0 to 16 ft. In total, 10 subsurface soil samples had detectable PCE concentrations, and 6 samples exceeded the 2006 Missouri Risk-Based Corrective Action Migration to Groundwater Risk-Based Target Level (11.8 and 52.7 mg/kg for residential and non-residential use for PCE; Jackson, 2009). No groundwater was encountered during any sampling events. Trichloroethene (TCE) and *cis*-1,2-dichloroethene (*cis*-DCE) were also detected in samples. The localized PCE detections in surface and subsurface soil in the field northeast of the building suggested that the former hat factory was a source location and, with the testimony from the Vienna water system operator, led the MDNR to speculate that perhaps this area was a former wastewater lagoon before the facility was connected to the city sewer system.

The MDNR also collected 19 tree-core samples and 13 shrub samples at the former hat factory and the surrounding area in October 2007 (fig. 1-06). Trees were targeted if they were most likely at suspected locations of contamination such as near building doors, loading docks, outfall pipes, or former lagoons (Kenneth Hannon, Missouri Department of Natural Resources, written commun., October 15, 2007). All samples had concentrations below the detection level for VOCs. This lack of detections may have been caused by heavy rains during the 3 weeks before sampling (Jackson, 2009), the time of the year that sampling was completed because VOC concentrations in trees have decreased more than an order of magnitude in fall and winter months (Limmer and others, 2014), or low-sensitivity methods used in measuring VOCs in tree-core samples because concentrations are often very low (on the order of nanograms per liter) in pore water. Tree-core sampling completed as part of

the RI indicated detectable concentrations of VOCs across the Vienna Wells site (see Section 3.1 of this report).

Figure 1-06. Locations of tree-core samples collected by Missouri Department of Natural Resources staff on October 16, 2007, at the Vienna Wells site, Vienna, Missouri. Modified from Jackson, 2009.

The MDNR SI also included the collection of water samples during February 2007, April 2007, September 2007, and October 2008. During February 2007, samples were collected from water taps at the Vienna City Hall and WOW Pizza to characterize VOC concentrations in the Vienna water distribution system at the point of exposure to the public (fig. 1-07). Concentrations of PCE at the Vienna City Hall were 0.45 µg/L immediately after the tap was turned on and 3.85 µg/L after running the tap for 5 minutes. Concentrations of PCE at WOW Pizza initially were 1.61 µg/L and decreased to 1.40 µg/L after running the tap for 5 minutes. During April 2007, 23 domestic water wells were sampled within 3 mi of the former hat factory (Jackson, 2009; fig. 1-07) ranging from 80 to 320 ft deep and constructed between 1957 and 1995. The wells were sampled after opening the faucet nearest the well and allowing water to flow for 5 minutes. None of the samples from the domestic wells contained measurable concentrations of PCE.

Based on the data collected during the SI, the site was scored and was considered for placement on the NPL solely because of its potential for migration to groundwater (USEPA, 2010). On September 27, 2010, the Vienna Wells site was added to the NPL because of detection of PCE above the MCL in well COV-03 (Missouri Department of Health and Senior Services, 2014).

Figure 1-07. Locations of domestic well and distribution samples collected in April 2007 and February 2007, respectively, Vienna, Missouri. Modified from Jackson, 2009.

1.4 Site Description

The Vienna Wells site consists of the former hat factory and the three public-supply wells COV-01, COV-02, and COV-03 (fig. 1-02). The 7.9-acre former hat factory property is in the northeast part of Vienna about 500 ft from public-supply well COV-03 and near several residential properties (figs. 1-04 and 1-08). The

footprint of the factory building is about 1 acre and is shaped like an "L" with sections extending north and east. The remaining area is about 6.9 acres and is mainly grassy and wooded areas. Surface topography slopes downward to the north, east, and south where adjacent lands are used for grazing of cattle. Ephemeral streams (North Tributary and East Tributary) bound the former hat factory to the north and east (fig. 1-02), and all runoff at the former hat factory is transported along these two ephemeral streams to Indian Creek, which ultimately empties into the Gasconade River. No known public water-supply intakes exist within 15 mi downstream from the former hat factory (Jackson, 2009).

Three small dumps are in the wooded area on the property (fig. 1-08), referred to in this report as the "central dump," "eastern dump," and "metal dump." Large metal debris, roofing material, old drums, bricks, and various metal cans are scattered in the central dump area to a depth of as much as 4 ft. The metal dump contains buried and partially buried metal debris in a roughly rectangular area about 150×30 ft to a depth of about 3 ft. The eastern dump is on the east side of the property, and debris in this dump area mostly consisted of old fur and occasional leather strips that ranged from a few inches to about 2 ft thick. Material in the central and eastern dumps seems to have been deposited on the surface, and material in the metal dump seems to have been deposited in an excavation.

Figure 1-08. Approximate location of dumps at the former hat factory property, Vienna, Missouri.

1.4.1 Soils

Soils at the Vienna Wells site are characterized as Gravois-Gatewood silt loam with clay content ranging from 12 to 80 percent (Jackson, 2009). Residuum, subsoil that forms as a result of weathered carbonate rock, below the soil is cherty silty clay that grades into cherty silt clayey gravel with depth (Jackson, 2009). The Gravois-Gatewood silt loam complex has a vertical hydraulic conductivity (K) of 1.19 to 39.7 feet per day (ft/d; Jackson, 2009). Soil and residuum range from 5 to 25 ft below ground surface (bgs).

1.5 Site History and Layout and Reported Chemical Use or Disposal

The former hat factory was operational from 1952 to 1996 (table 1-01; appendix A, figs. A-01 through A-

03). During that time, the facility was owned by two separate companies and at least two solvents (PCE and 1,1,2-trichloro-trifluoroethane [CFC-113]) were apparently used. Oral history was obtained from employees that worked at the facility from about 1959 through 1969. As of 1959, the facility only shaped what are called "hat bodies" made from wool into finished hats using a combination of metal dies and wood blocks to size and shape the hats. The wool hat bodies were cone-shaped wool "felt" that were precleaned and dyed when they arrived. The first step was to "size" the hat bodies, which was done using metal dies to make the appropriate size of the hat. Then "blocking" was done whereby the hat shape was made by pressing the sized hat body into the desired shape using wood "blocks." A small steam press was used in these operations. After blocking, the hats moved to the sewing area where the brims were cut and rolled into the desired shape, and silk liners and sweat bands were installed. Finished hats were taken to the warehouse area when they were boxed for shipment. PCE may have been used as a "spot" remover to clean finished hats, as was documented at another Langenberg hat factory at the New Haven Superfund site (USEPA, 2006). It is assumed that operations from 1952 to 1959 were similar, and the primary changes included expansions of the original building west of the office (restrooms, maintenance and parts shop, and boiler room) and east of the office (additional storage of hat bodies; fig. 1-09).

Table 1-01. General operational history of the former hat factory, Vienna, Missouri.

Figure 1-09. Operational history and locations of former process areas and general dates of building additions of the former hat factory, Vienna, Missouri.

Sometime in the early to mid-1960s, the facility began receiving some raw wool that was "corded" or spun into a fabric that was used to make hat bodies. Cording/spinning was originally done in the west part of the shipping area. Shortly thereafter, a major expansion added about 40,000 square feet to the existing building and included the dye "forming" room (room with the channels in the concrete floor) and the two-story, north extension of the "L" shape in 1965-67 (fig. 1-09). This expansion was done to allow the facility to begin processing raw fur: mostly rabbit-fur hides into finished rabbit-fur hats. Rabbit hides were received in 500-pound (lb) bales and stored in the upper floor of the north "L" extension. Bales were loaded from small trucks through

doors to the west or from tractor-trailer rigs from a loading dock on the east side of the "L" extension. The hides were packed in wood crates with mothballs, and former employees noted the strong smell of mothballs in this area. The fur was removed from the hides in the basement of the northern "L" extension using a complex series of mechanical devices that "removed the hide from the fur" according to former workers. Former employee accounts also indicate the use of acid baths during fur processing in the northeast area of the basement. It is not known whether these acid baths happened before or after cleaning of natural oils from the raw fur where solvent use was likely. The process from the raw fur to the finishing area is not clear, but the fur was formed into the initial hat bodies on three large "forming machines." Three circular holes in the slab on the west side of the building mark the location of these machines (fig. 1-09). The raw fur was cleaned and dyed before it was "formed" into hat bodies in the forming room, but details of this process are not clear. What is clear is that large quantities of steam and water were used in this process, and the condensed steam and water from dying and forming was channeled out of the area through the troughs in the floor and ran out the west side of the building. At a similar Superfund site in New Haven, Missouri, sulfuric acid was used in the dying process to "set" the dyes; if sulfuric acid was used at the former hat factory, it would account for the poor quality of the concrete floors in this area. Some former employees recall a sweet solvent smell and a large machine that sat on a raised concrete pad in the dye and forming room that may have tumbled fur. A final addition on the south side of the dye room was done sometime between 1969 and 1996. It is not known what operations took place inside this addition (fig. 1-09).

Sometime in the 1980s the facility transitioned from manufacturing fur hats to ladies hats and finally to ball caps and cowboy hats, and the company eventually changed to "Top This, Inc." based in New York. Top This Inc. was owned by two sisters, and when the principle owner died shortly after the ownership change, the remaining owner was unable to run the business, so the facility closed in the 1990s. Top This, Inc. dissolved in 1997, which was followed shortly after by the dissolution of the Langenberg Corporation in 2000 (Jackson, 2009).

There is no evidence that solvents were used in the finishing area in the east one-half of the building (shipping, sewing area, hat press room, hat-body storage, restrooms, and office). Solvents probably were used in

the maintenance area that was between the boiler room and the sewing area and in the dying-forming room areas or maybe the basement of the north extension. The largest amount of solvents probably was used in the southwest part of the facility to clean raw fur either before or after the forming of hat bodies. The MDNR noted up to eight empty 35-gallon drums of Valclene[®] inside the building during 2007 (Jackson, 2009), and the USGS observed several more in dumps around the property. Valclene[®], a solvent containing CFC-113, was introduced by DuPont in 1964 (Johnson, 1971). The vapor pressure of CFC-113 is nearly 20 times that of PCE and allowed laundry to be dried at lower temperatures and was promoted as the preferred dry-cleaning solvent for delicate fabrics and fur (Linn, 2009). Valclene[®] was never widely used by dry cleaners, production was banned in 1996 (United Nations Environment Programme, 1987), and it is no longer in use (Linn, 2009). Also, detection in the environment is difficult because mass transfer is rapid from the sorbed and dissolved phase to the vapor phase because of the high volatility of CFC-113. It is unknown what solvent was used at the former hat factory from 1952 to the introduction of Valclene® in 1964. There is not direct knowledge of PCE being used at the facility.

Since its closure in the late 1990s, the former hat factory has deteriorated and is currently partially demolished leaving mostly exposed floor slabs and partially collapsed roofs (fig. 1-09). The only remaining intact roof is on the northern "L" extension and above the former office. The roof on the eastern part of the factory is failing and allowing stormwater to pass through. A floor drainage system in the dye area at the southwestern corner of the facility collects stormwater from the surrounding impervious area and redirects it to the ditch directly west of the factory. An open, concrete-lined culvert along the southwest perimeter of the building directs water to the same ditch. Areas south of the former hat factory were historically gravel lots used for employee parking (appendix A, figs. A-01 through A-03). Several doors are around the perimeter of the former hat factory, including loading docks on the north and east sides of the northern extension of the building. A septic system is on the north side of the former hat factory, as well as sewer lines running from the south and north sides of the facility.

1.6 Physical Characteristics of the Vienna Wells Site

1.6.1 Demography, Land use, and Wildlife

Vienna has a population of 610 (U.S. Census Bureau, 2010a) and an incorporated area of about 1.1 square miles. Vienna has a median age of 47.3 years and an average household size of 2.1. About 43 percent of homes are owner-occupied, 34 percent are rented, and about 23 percent are vacant. Vienna's land use is mixed urban and rural with much of the rural area classified as cropland or deciduous forest (Missouri Spatial Data Information Service, 2005). The areas directly outside of Vienna are almost exclusively deciduous forest and cropland. The mixed urban area predominantly consists of mid-1900s era, single-family residences and commercial facilities but also consists of some multifamily residences and several churches. The downtown area of Vienna, which is on the southeast corner of the intersection of U.S. Highway 63 and Missouri Route 42, is small and consists of about four blocks of small businesses and the Maries County courthouse. The main employer in Vienna is the Maries Manor Nursing Home, which employs about 63 people (Meramec Regional Planning Commission, 2014).

No Federal or State endangered or threatened animal species are known to be present on the Vienna Wells site (Missouri Department of Conservation, 2009). A total of 12 State-listed and or Federal-listed endangered species identified during the ERA may be present in Maries County, which include the following:

- Crystal Darter (*Crystallaria asprella*) – fish (State "Endangered")

- Eastern Hellbender (*Cryptobranchus alleganiensis alleganiensis*) – salamander (State "Endangered")

- Ebonyshell (*Fusconaia ebena*) – mussel (State "Endangered")

- Elephantear (*Elliptio crassidens*) – mussel (State "Endangered")

- Gray Bat (*Myotis grisescens*) (State and Federal "Endangered")

- Indiana Bat (*Myotis sodalis*) (Federal "Endangered")

- Northern Long-eared Bat (*Myotis septentrionalis*) (proposed Federal "Endangered")

- Niangua Darter (*Etheostoma nianguae*) – fish (State "Endangered", Federal "Threatened")

- Pink Mucket (*Lampsilis abrupta*) – mussel (State and Federal "Endangered")

- Running Buffalo Clover (*Trifolium stoloniferum*) (State and Federal "Endangered")

- Scaleshell (*Leptodea leptodon*) – mussel (State and Federal "Endangered")

- Spectaclecase (*Margaritifera monodonta*) – mussel (Federal "Endangered")

1.6.2 Surface Features

The town lies across a topographic divide between the Gasconade River about 2.0 mi to the east and the Maries River about 1.9 mi to the west-northwest (fig. 1-01). The town is in the Salem Plateau physiographic subprovince of the Ozark Plateaus Province (Fenneman, 1938). The Salem Plateau is characterized by a moderate to rugged terrain with thin soils and narrow steep-walled valleys (Imes and Emmett, 1994). Topographic relief is the result of gradual uplift of the Ozark Dome in southern Missouri and erosion of the uplifted rocks by precipitation, runoff, and streamflow (Imes and Emmett, 1994). The topographic relief in the Vienna area is accentuated because of its proximity to the Gasconade River, which controls the base level for streams in the region. Land-surface elevation ranges from 615 ft above the NAVD 88 along the Gasconade River to about 900 ft on a hilltop within the central part of the town. Fly Creek, a tributary to the Maries River with an elevation of about 760 ft, is less than 1.0 mi west of town (fig. 1-01), and Indian Creek, a tributary to the Gasconade River with an elevation of about 710 ft, is directly east of town (fig. 1-02). The region is generally rural and consists of gently rolling hills with farmland (pasture) and scattered deciduous forest.

1.6.3 Climate

Vienna has a temperate climate with hot summers and moderately cool winters. The mean annual precipitation in Vienna is about 45 inches (in.) with about 75 percent falling from April through November (National Oceanic and Atmospheric Administration [NOAA], 2015). The 2-year, 24-hour average rainfall is about 3.38 in. (NOAA, 1992). The mean annual temperature is about 57.5 degrees Fahrenheit (°F). From January 2000 through January 2015, the recorded temperature extremes in Vienna were a minimum of -14.0 °F (January 2001) and a maximum of 109 °F (August 2007). During 2012, Missouri weather included the worst drought in 30 years, which lead to the designation of all Missouri's 114 counties as Primary Natural Disaster areas by the U.S.

Secretary of Agriculture (U.S. Department of Agriculture, 2012).

1.7 Surface-Water Hydrology

Vienna lies across a topographical divide between the Gasconade River and the Maries River (fig. 1-01), and two smaller tributaries, Fly Creek (fig. 1-01) and Indian Creek (fig. 1-02), transport surface-water runoff in the Vienna area to the west and east to the Maries River and the Gasconade River, respectively. The Gasconade River is 271 mi long, empties into the Missouri River, and is fed by a number of springs because of the karst geology of the Roubidoux and Gasconade Dolomite Formations. The Gasconade River controls the base level for most streams in the region and has a gradient of 27.6 and 3.9 feet per mile (ft/mi) in the upper and lower regions, respectively (Blanc and others, 2001). The Maries River is 41.5 mi long and empties into the Osage River about 10 mi upstream from the confluence of the Osage River and Missouri River (Schubert, 2001).

1.8 Regional Hydrogeology

The main aquifer in the Vienna area is the Ozark aquifer, which supplies all domestic, industrial, and public water used in the Vienna area.

1.8.1 Ozark Aquifer and St. Francois Confining Unit

Vienna is underlain by the Ozark aquifer, which is about 1,000 ft thick (Miller and Vandike, 1997). The Ozark aquifer is a thick sequence of water-bearing dolostone, limestone, and sandstone formations ranging in age from Late Cambrian to Middle Devonian (Imes and Emmett, 1994). Although these units collectively are a regional aquifer, the water-yielding capacity of the various individual units is variable (table 1-02). In the Vienna area, the Ozark aquifer consists of the Jefferson City Dolomite, Roubidoux Formation, Gasconade Dolomite (including the basal Gunter Sandstone Member), Eminence Dolomite, and the Potosi Dolomite (table 1-02). The Ozark aquifer is underlain by the St. Francois confining unit, which is composed of the Derby-Doe Run Dolomite. Geologic names in this report conform to terminology used by the MDNR Division of Geology and Land Survey (Miller and Vandike, 1997).

Table 1-02. Geological unit properties in the Vienna Wells site area, Vienna, Missouri.

The uppermost bedrock unit varies with surface topography in the Vienna area and includes the Jefferson City Dolomite, Roubidoux Formation, and Gasconade Dolomite (MDNR, 2007). The Jefferson City Dolomite, where present, ranges from 20 ft to a maximum of about 70 ft in thickness north of Vienna, is a tan to light gray, fine- to medium-crystalline dolostone (equivalent to dolomite), or argillaceous dolostone (Thompson, 1991). The unit contains thin greenish-gray mudstone and shale beds and several chert-rich zones. Overall, the Jefferson City Dolomite is a poor water-producing formation and typically has low vertical and horizontal K values. Where the Jefferson City Dolomite is exposed at the surface, vertical infiltration of precipitation can be impeded more than in older formations (Miller and Vandike, 1997).

The Roubidoux Formation underlies the Jefferson City Dolomite and is present at the land surface in the vicinity of the former hat factory. Regionally, the lithology of the Roubidoux Formation is variable and includes sandstone, sandy dolostone, dolostone, mudstone, chert, and cherty dolostone (Thompson, 1991). Where the Roubidoux Formation is present in the Vienna area, the unit is about 25 to 170 ft thick (appendix E, fig. E-01). It can be distinguished from the overlying Jefferson City Dolomite by an abrupt increase in chert (20 percent or more) at the top of the formation. Weathered surfaces range in color from brownish-gray to yellowish-orange, and unweathered surfaces appear creamy-tan to white (Miller and Vandike, 1997). Regionally, the Roubidoux Formation yields 15 to 35 gallons per minute (gpm) where shallow and 50 to 75 gpm where deeply buried (Miller and Vandike, 1997). The K values of the Roubidoux Formation have only been measured in Arkansas where they range from 0.02 to 1.76 ft/d (Pugh, 2008). Although the Roubidoux Formation is cased out in the city public-supply wells, many of the domestic wells in Vienna are open to the Roubidoux Formation. Most on-site monitoring wells installed during the RI (wells MW-01, MW-02, MW-03, MW-04, MW-05, MW-06, MW-07, and MW-09) are open to the Roubidoux Formation.

The Gasconade Dolomite, which underlies the Roubidoux Formation, has an average thickness in the Vienna area of about 260 ft (appendix E, fig. E-01). The Gasconade Dolomite is divided into two informal units (upper and lower Gasconade Dolomite) and the basal Gunter Sandstone Member. The upper Gasconade Dolomite generally is a massively bedded, medium crystalline, light gray dolostone with small amounts of chert and

sandstone and is about 20 to 70 ft thick in Vienna. The upper Gasconade Dolomite tends to be less permeable than the overlying Roubidoux Formation or the underlying lower Gasconade Dolomite. Because of this lower permeability, the surface casing in many large-capacity production wells, including city public-supply wells COV-01, COV-02, and COV-03, is set near or at the bottom of this unit (Miller and Vandike, 1997; figs. E-01, 3-05, and 3-06). The lower Gasconade Dolomite contains abundant chert, some of which forms massive chert beds 10 to 20 ft thick, and is light-brownish gray in color. Yields from wells open to both the upper and lower Gasconade Dolomite generally range from 10 to 20 gpm. The Gunter Sandstone Member is the basal unit of the Gasconade Dolomite, ranges from 20 to 35 ft thick in the Vienna area, and is a target zone for many large-capacity wells in Missouri including Vienna public-supply wells. Although referred to as a sandstone, the Gunter Sandstone Member contains less than 10 percent sand according to geologic logs from the Vienna area. Wells open to the Gunter Sandstone Member in the region typically yield 20 to 70 gpm. The K values of the Gunter Sandstone Member have only been measured in Arkansas where they range from 0.01 to 1.90 ft/d (Pugh, 2008).

The Eminence Dolomite, Potosi Dolomite, and Derby-Doe Run Dolomite are the oldest geologic units used by wells in the Vienna area and are primarily accessed by public-supply wells (appendix E, fig. E-01). The Eminence Dolomite, which underlies the Gasconade Dolomite, is a medium- to coarsely-crystalline dolostone with little or no chert. Well logs indicate that the Eminence Dolomite is about 240 ft thick in the Vienna area. The Potosi Dolomite is the lowermost geologic unit in the Ozark aquifer and consists primarily of massive to thickly bedded "vuggy" dolostone with abundant drusy quartz (Miller and Vandike, 1997). The Potosi Dolomite is about 200 ft thick in the Salem Plateau and about 315 ft thick in the Vienna area. The K values of the Potosi Dolomite have only been measured in Arkansas where they range from 0.12 to 1.78 ft/d (Pugh, 2008). The lower Gasconade Dolomite and underlying Eminence and Potosi Dolomites are important sources of water for large-capacity wells in Vienna and throughout much of southern Missouri. Collective yields from these units to wells are typically in the range of hundreds of gallons per minute or more. Regionally, the Potosi Dolomite is the most permeable and most reliable source of water in the Ozark aquifer for large-capacity wells in southern Missouri (Imes and Emmett, 1994).   The Derby-Doe Run Dolomite is composed of shale, siltstone, fine-grained sandstone, dolostone and limestone conglomerate and has low chert content (Miller and Vandike, 1997). Public-

supply well COV-03 penetrates into the Derby-Doe Run Dolomite, the upper formation of the St. Francois confining unit, which yields 30 to 50 gpm in upper sections (Miller and Vandike, 1997).

1.8.2 Geologic Structure

No known geologic structures exist within a 10-mi radius of Vienna, and bedrock units are essentially horizontal. The closest geologic structure is a 13-mi-long fault more than 10 mi to the southwest and extends to the northwest into Miller County. Other geologic structures outside of the Vienna area include a 10-mi-long anticline and 7-mi-long fault about 13 mi to the southeast (MDNR, 2007).

1.8.3 Groundwater Flow

Groundwater in the Ozark aquifer generally is unconfined throughout the Salem Plateau (Imes and Emmett, 1994). Regional groundwater movement generally is from upland areas between major rivers and streams toward valleys where groundwater discharges as base flow to the streams. The regional flow system generally is present in the deeper parts of the aquifer (Potosi Dolomite), except near regional recharge or discharge areas where flow enters or leaves the aquifer. Regional groundwater flow in the Vienna area is northeast toward the Gasconade River (MDNR, 2007; fig. 1-10). Horizontal K values ranging from 0.10 to 5.0 ft/d (Czarnecki and others, 2009; Richards, 2010; Imes, 1989) and vertical K values ranging from about 1×10^{-3} to 1×10^{-1} ft/d (Richards, 2010; Imes, 1989) have been used in modeling groundwater flow in the full thickness of the Ozark aquifer.

Figure 1-10. Regional potentiometric map of Vienna and the surrounding area developed from driller's logs from 1987 to 2005 by the Missouri Department of Natural Resources, Missouri (from MDNR, 2007).

2.0 Remedial Investigation Activities at the Vienna Wells Site

A chronological summary of major RI activities is given in table 2-01. Early RI activities at the Vienna Wells site focused on reconnaissance sampling of all creeks, seeps, and sewer systems and a review of historical aerial photography looking for areas of land disturbance at the former hat factory property consistent with

possible dumping of wastes (appendix A, figs. A-01, A-02, and A-03). A more detailed reconnaissance at the

Vienna Wells site included tree-core sampling, a metal detector survey, soil borings, and groundwater sampling.

Several monitoring wells were installed in the vicinity of the former hat factory to accurately describe the water-

table surface, delineate PCE concentrations, and determine the connection between PCE in well COV-03 and

groundwater at the former hat factory property. Results from the tree-core reconnaissance were used to design the

soil sampling effort, which included the collection of surficial and subsurface soil samples. Excluding tree-core

samples, which were analyzed by the Environmental Research Center at the Missouri University of Science and

Technology, all samples were typically analyzed by the portable gas chromatograph (GC) and screened for VOCs

with a subset submitted for lab analyses (appendix B, tables B-01 through B-07). All laboratory water, soil, and

sediment samples collected by the USGS were analyzed by Test America Laboratories, Inc., the USGS National

Water-Quality Laboratory in Denver, Colorado, or by RTI Laboratories, Inc., in Livonia, Michigan. Samples that

were collected as part of the HHRA were analyzed for constituents on the hazardous substance list (HSL),

referred to as HSL samples in this report, which included VOCs, semivolatile organic compounds (SVOCs),

organochlorine pesticides, polychlorinated biphenyls, organophosphorus pesticides, inorganic anions, metals, and

percent-moisture (appendix K, table K-01). All soil and vapor-intrusion samples collected by the USEPA were

analyzed by the USEPA Region 7 laboratory in Kansas City, Missouri, for a similar list of constituents (table K-

02).

The following sections discuss the environmental media investigated and general approach to sampling

these media for the Vienna Wells site RI. The sections are organized by those activities that generally were done

first or of a site-wide reconnaissance nature, such as tree-core sampling and sampling of surface water and

springs, followed by the groundwater and soil investigations. Additional ancillary activities also discussed include

a reconnaissance sampling of sanitary sewers, borehole geophysical surveys, sampling of the Vienna public water

supply distribution system, and vapor-intrusion sampling by the USEPA, followed by a general overview of

analytical methods used in the investigation.

Table 2-01. Summary of Remedial Investigation activities at the Vienna Wells site, Vienna, Missouri, 2011-2016.

2.1 Tree-Core Reconnaissance

Volatile organic compound concentrations were measured in water collected from cores taken from 488 trees at and near the Vienna Wells site. Tree-core sampling was completed during summer 2011, spring 2013, summer 2013, summer 2014, spring 2015, and fall 2015 (fig. 2-01; appendix B, table B-05) following the project Quality Assurance Project Plan (QAPP; USEPA, 2011b). Large trees (diameter greater or equal to 6 in.) were targeted, although many small trees (diameter less than 6 in.) were sampled when large trees were not available. No specific tree species were targeted; however, cedar trees were avoided because natural compounds in cedar-tree tissue coellute with toluene.

Figure 2-01. Locations of tree-core samples at the Vienna Wells site, Vienna, Missouri, 2011-2015.

2.2 Stream, Spring, and Seep Sampling

Water samples were collected during September 2011, May 2012, and March 2014 during seepage runs to quantify contributions to surface flow and associated constituents. The first and second collections included stream discharge measurements, measurement of water specific conductance and temperature, and collection of VOC screening samples from various seeps in the vicinity of Vienna. The purpose of these two sample collections was to screen for VOC contaminants in nearby streams, springs, and seeps during base-flow and high-flow conditions. The third sample collection included the collection of spring and seep samples from the East Tributary near the former hat factory property. Samples collected during this third sample collection were used in the BRA.

During the September 2011 and May 2012 sample collections, flow observations were made at 149 sites (47 main stem stream sites, 97 tributary sites to these streams, and 5 springs or seeps; appendix B, table B-08; figs. 2-02 and 2-03), and water samples were collected at 62 sites.

Figure 2-02. Locations of discharge measurements during the seepage run in September 2011, Vienna, Missouri.

Figure 2-03. Locations of discharge measurements during the seepage run in May 2012, Vienna, Missouri.

During September 26 to 29, 2011, stream discharge was measured in the vicinity of Vienna from the Gasconade River east to the divide and then east to and along Fly Creek (fig. 2-02). Measurements were taken during a period of minimum streamflow when stream discharge is sustained by diffuse groundwater and spring inflow and not by surface runoff. Discharge measurements were made following the methods described by Rantz and others (1982). Discharge was estimated at sites where it could not be measured either because of insufficient water depths, obstructions in the stream channel, or minimal flow velocities. All estimated discharges are rated poor, meaning the determined discharge value is likely greater than 8 percent of the true discharge. Specific conductance and water temperature generally were measured at the sites at the time of the discharge measurement to identify potential contamination sources and areas of large groundwater discharge indicated by anomalously large specific conductance or anomalously low temperature. Specific conductance was measured with a temperature-compensating meter calibrated to read in microsiemens per centimeter at 25 degrees Celsius, and water temperature was measured with a thermistor to the nearest 0.1 degree Celsius following procedures described by Wilde and Radtke (1998).

All the streams consisted of sand- and gravel-filled channels (streambed sediment) that include frequent bars composed of loose sand, gravel, and cobbles. Discharge measurements near large sand and gravel bars were avoided when possible. During low flow, a large percentage of water can travel through the streambed sediment instead of flowing on the surface. At several locations, the water flowed through the streambed sediment and was temporarily lost to surface flow but resurfaced farther downstream. In these instances, flow is not lost to the underlying bedrock aquifer; therefore, these stream reaches were not identified as losing streams.

Discharge measurements also were measured on Indian Creek to the east of Highway 63 and the East Tributary on May 8, 2012. The intent of these measurements was to collect a concentrated set of data in the vicinity of the potential contamination source, the former hat factory property, during high base flow conditions. Although the base flow in the Gasconade and Bourbeuse Rivers was larger than during the September 2011 sample collection, the area was experiencing below normal precipitation (USGS, 2012), and antecedent conditions were very dry; therefore, groundwater discharge was likely low. About 0.35 in. of precipitation fell on May 7, 2012, before the samples were collected (NOAA, 2015). Although this may have dampened the soils, no

runoff was observed. Discharge was measured or observed at 32 sites (fig. 2-03; appendix B, table B-08).

The North and East tributaries were investigated during the September 2011 and May 2012 sample collections from the headwaters to where the East Tributary intersects with Indian Creek south of Highway 42 (figs. 2-02 and 2-03). No flow was observed in the North Tributary (site WP-003); however, flow was observed in the East Tributary beginning at site WP-012 and ceasing in the vicinity of site WP-018 (fig. 2-03). Because the flow in the East Tributary was small where it was flowing (maximum flow measurement of 0.06 cubic foot per second at site WP-017), it is difficult to determine if the lost flow was truly a losing stream reach or if flow was moving through the gravel. The creek continued to be predominantly dry, with scattered pools and occasional short-duration stream lengths with minimal flow to site WP-021 until site PB-588 where flow increased to a maximum of 0.009 cubic foot per second directly upstream from Indian Creek. Downstream from site M-09, flow continued to gradually increase in the tributary, especially south of Highway 42, to its mouth at the Gasconade River (fig. 2-02).

Between the two measurement events in September 2011 and May 2012, water-quality samples were collected and analyzed for PCE and TCE at 81 sites, including 12 small springs or seeps (appendix D, fig. D-09). Spring site WP-020 was not observed during the September 2011 seepage run but was located in February 2012.

The third surface-water sample collection took place on March 3, 2014, with the objective to collect samples for laboratory analyses (hereinafter referred to as laboratory samples) from the North and East Tributaries for the USEPA and HHRA. About 0.35 in. of precipitation fell on March 2, 2014, before the start of sampling (NOAA, 2015). Samples were collected from the spring WP-020 and three locations (sites TB2-1 and TB2-2, fig. 2-04; and TB2-Trickle, figs. 2-04 and D-09 [appendix D]) from the East Tributary. Site TB2-1 was upstream from the confluence of the North Tributary and East Tributary, site TB2-2 was downstream from the same confluence, and site TB2-Trickle was a seep from what seemed to be a bedrock outcrop in the East Tributary. Attempts were made to collect water samples from the North Tributary, but no pools were identified. Additionally, a small spring (site JW-SEEP) was identified southeast of the former hat factory property (fig. 2-04), and a sample was collected from the mouth of the spring for the portable GC. Subsequently, this site was added to the quarterly monitoring schedule. Streambed sediment samples were collected from three sites on the

East Tributary (sites WP-020, TB2-1, and TB2-2), one site on the North Tributary (site TB1-2), and one site along the ditch west of the former hat factory (site WD-01; fig. 2-04).

Figure 2-04. Locations of volatile organic compound (VOC) samples, hazardous substance list (HSL) samples, and sewer and septic drains at the Vienna Wells site, Vienna, Missouri.

2.3 Groundwater Investigation

Between the fall of 2011 and the spring of 2016, a total of 116 groundwater laboratory samples were collected from public-supply, domestic, and monitoring wells (table 2-02; well construction diagrams are provided in appendix H, figs. H-01 though H-09) using methods described in the QAPP (USEPA, 2011b). Briefly, samples were collected from either a submersible pump lowered to the open interval of the monitoring well (table 2-02) or from the dedicated pump in the well. In addition, 302 groundwater samples were analyzed by the portable GC (appendix B, tables B-01 and B-07). Domestic and public-supply well construction data were obtained from the well owner when known or from the MDNR well construction database (MDNR, 2007).

Table 2-02. Summary of bedrock and overburden monitoring wells at the Vienna Wells site, Vienna, Missouri.

Measured water levels in monitoring, domestic, and public-supply wells were used to develop potentiometric maps in the vicinity of the former hat factory property (figs. 2-05 and 2-06). Groundwater levels in monitoring and domestic wells were measured from the top of the well casing using a calibrated electric tape and read to the nearest 0.01 ft. The length of the well casing that extended above land surface was measured to the nearest 0.1 ft and was subtracted from the water-level measurement to reference the water level to land surface. Groundwater levels in public-supply wells were measured to the nearest foot using calibrated airlines installed in each public-supply well. The depth of the submerged end of the airline was subtracted from the measured height of water.

Figure 2-05. Potentiometric map of the Ozark aquifer in the Vienna, Missouri, area, 2011 and 2016.

Figure 2-06. Potentiometric map of the former hat factory property and surrounding area and topographic

contours in Vienna, Missouri, 2011 and 2016.

Land-surface elevations of monitoring wells used in the development of the potentiometric map in the vicinity of the former hat factory property (fig. 2-06) were obtained from differential leveling to an accuracy of 0.1 ft. Based on the accuracy of the various measurements used to determine the water-level elevation, the largest source of error in the potentiometric map in the vicinity of the former hat factory is in the measurement of the length of the well casing above land surface.

2.3.1 Domestic Wells

Water levels were measured in 45 domestic wells (appendix D, table D-03), and water samples were collected from 54 wells within about 2 mi of the former hat factory property during November 15 to 30, 2011 (appendix B, table B-01). Only one known privately owned well (well MK-01) is within the Vienna city limits (Shon Westait, Vienna City Utilities manager, oral commun., 2011), and most of the inventoried domestic wells outside the city limits were concentrated along major roads. Because of the lack of wells, water-level data are scarce in the Vienna city limits, so groundwater level information in this area was supplemented with surface-water level data to construct a detailed potentiometric map of the Vienna area. After a water-level measurement was obtained, wells were purged until specific conductance, temperature, and pH stabilized and water samples were collected according to the QAPP (USEPA, 2011b). Water-level measurement data from domestic wells and surface-water features in 2011 and water-level measurement data from public-supply wells COV-01 and COV-03, monitoring well MW-08, and domestic well MK-01 in 2016 were used to construct a potentiometric map of the Vienna area (fig. 2-05). Land-surface elevations for the potentiometric map of the Vienna area (fig. 2-05) were obtained from USGS digital elevation model data, which were derived from topographic contour maps with a 20-ft contour interval, which implies that the elevations are accurate to plus or minus 10 ft, or one-half the contour interval. Water-level data collected in 2016 were used to refine the contours in the vicinity of the former hat factory property to reflect depression in the potentiometric surface from the pumping of public-supply well COV-03. After public-supply well COV-03 had been pumping for more than 2 hours, water levels were measured

in public-supply wells COV-01 and COV-03, monitoring well MW-08, and domestic well MK-01. Because

domestic well MK-01 is more than 1,000 ft from public-supply well COV-03, it was assumed that the depression

caused by the pumping of public-supply well COV-03 would be minimal in the vicinity of domestic well MK-01.

Water levels were measured in domestic well MK-01 in 2011 and 2016, and the water table in domestic well

MK-01 in 2011 (765.60 ft bgs) was about 11 ft lower than in 2016 (777.01 ft bgs). These two sets of water-level

data were used to approximate the potentiometric surface in the vicinity of the former hat factory property under

pumping conditions imposed by public-supply well COV-03 (figs. 2-05 and 2-06). Although uncertainties are

introduced when comparing water levels measured 5 years apart and with different completion intervals, the

magnitude of the depression in the potentiometric surface caused by public-supply well COV-03 is much larger

than any errors introduced by differences in time and completion interval, as well as errors introduced from

contour maps.

On March 31, 2014, a subset of the wells sampled during November 2011 and two additional wells (wells

VIN-PB-01 and VIN-PB-02) were sampled in support of the HHRA. A total of seven laboratory samples were

collected from wells MK-12, MK-29, MK-34, MK-35, MK-43, PB-01, and PB-02. Laboratory samples collected

from domestic wells in March 2014 were analyzed only for VOCs.

In June 2014, the existence of a hand-dug cistern (site JW-01) was discovered to the southeast of the

home about 200-ft south of the former hat factory property. The cistern is 18.2 ft deep and lined with smooth

rock. Starting in September 2014, water samples were collected from site JW-01 during every quarterly

groundwater monitoring event.

2.3.2 Monitoring Wells

Based on the tree-core results and knowledge about PCE concentrations in domestic well MK-01 south of

the former hat factory, monitoring wells were installed first on the former hat factory property from May 2012 to

December 2014 and then off site from June 2015 to March 2016. With the exception of monitoring well MW-08,

monitoring wells were installed using a cable-tool drill rig rather than the quicker air-rotary method commonly

used for installation of wells in the region. Cable-tool drilling injects no high-pressure air or water into the

borehole and results in samples that are more quickly representative of ambient conditions in the aquifer at the depth sampled as compared to other drilling methods. In cable-tool drilling, a weighted bit (1,000 to 2,000 lbs) suspended on a cable is dropped repeatedly from a height of about 2 ft to break the bedrock. The bit is removed about every 5 ft of depth and a bottom-filling steel bailer is used to remove the broken rock and water slurry from the bottom of the borehole and transfer it to a container. A sample of the drill cuttings, a mixture of water and rock, discharged from the bottom of the first bailer removed from the borehole after each drill cutting removal depth (about every 5 ft) was collected in a standard VOC vial and analyzed by the portable GC (appendix B, table B-03). Nearly continuous profiles of PCE concentrations in each borehole were established based on a total of 394 samples and used in combination with mid- and post-drilling water-level measurements and potentiometric maps to guide the depth of the open interval of each monitoring well. Drill-cutting samples were collected during air-rotary drilling of well MW-08 about every 10 ft by placing a standard VOC vial slightly into the edge of the stream of cuttings discharged from the well casing as the cuttings were diverted from the top of the borehole into a roll-off container. Water samples also were collected from the borehole at the beginning or end of each day as the water accumulated in the borehole (appendix B, table B-01; labeled with feature code "BH") and after the completion of the well (table B-01; labeled with feature code "MW").

Most bedrock monitoring wells had two open intervals in the same borehole: a deeper interval and a shallower interval. These clustered wells were constructed with an annular space between the surface casing and the riser of the deeper well (for example, wells MW-01 and MW-01A; appendix H, fig. H-01). The deeper well consists of a screened interval of polyvinyl chloride (PVC) riser surrounded by a filter-sand pack. The PVC riser has a flange and rubber packers forming a grout cup at the bottom. The riser is suspended from the steel surface casing using a clamp and chain, and then a Portland cement or bentonite plug is installed above the grout cup through a tremie pipe. If cement is used, the annulus between the top of the cement plug and the bottom of the steel surface casing is partially filled by carefully pumping bentonite grout from the bottom of the annulus through the tremie pipe and monitoring the rising level of grout in the annulus. Several feet of filter-pack sand is placed on top of the bentonite slurry, and the remaining portion is open to the bottom of the casing. Although slightly more complex in design, this construction decreased the total number of boreholes drilled and avoided

complications caused by drilling monitoring wells in close proximity to one another. A summary of monitoring wells is provided in table 2-02.

Groundwater was collected according to the QAPP in the monitoring wells during January 2013, April and May 2013, July 2013, December 2013, March 2014, May 2014, September 2014, April 2015, and March 2016. A total of 79 laboratory samples were collected during these samplings. Water levels were measured in monitoring wells in the vicinity of the former hat factory property and used in conjunction with water-level measurements from public-supply well COV-03 to create a potentiometric map of shallow groundwater at the Vienna Wells site (appendix D, fig. D-01). Land-surface elevations of monitoring wells used in the development of the potentiometric map in the vicinity of the former hat factory property were obtained from differential leveling to an accuracy of 0.01 ft. Based on the accuracy of the various measurements used to determine the water-level elevation, the largest source of error in the potentiometric map in the vicinity of the former hat factory property is in the length of the well casing above land surface.

In addition to discrete water-level measurements, continuous water-level measurements were made using submersible pressure transducers installed in shallow wells MW-06A and MW-07 and deep well MW-06 (appendix D, fig. D-10). Additionally, data loggers were secured to electrical equipment in the well house of well COV-03 to record the timing of pumping events in well COV-03.

Slug tests were done on all monitoring wells to calculate the horizontal K value of the Ozark aquifer at the former hat factory property. Slug tests were performed from December 2015 to April 2016 in wells open to the Roubidoux Formation and Eminence Dolomite. Slug test methodology and findings are in appendix F at the back of this report.

2.3.3 Geophysical Logging

From January 5 to 6, 2012, the pump was removed from well MK-01; and geophysical logging was completed using caliper, hear-pulse flowmeter, natural gamma, fluid specific conductance, and fluid temperature logs (appendix G, fig. G-01). The caliper log indicated the bottom of the well casing was at 80 ft bgs, and the hole was of uniform diameter to the bottom of the well at about 148 ft, with the exception of a 2-ft-long section, about

9 in. in diameter, directly below the bottom of the casing. On August 17, 2015, and September 22, 2015, geophysical logging was completed on monitoring well MW-08 using caliper, heat-pulse flowmeter, natural gamma, fluid specific conductance, and fluid temperature logs under ambient (fig. G-02) and pumping (fig. G-03) conditions.

2.4 Soil Investigation

A total of 956 soil samples were collected by the USGS from 190 soil borings at the former hat factory property (fig. 2-07) and were analyzed by the portable GC (905 samples) or analytical laboratory (51 samples). Samples analyzed by the portable GC generally were collected at a depth of 1 to 2 ft in each boring. Soil borings were done during March to May 2014 at the former hat factory property and at one background site near well COV-03 (site SBG-01; fig. 2-07). Special focus was given to areas of interest based on PCE detections in tree-core samples and beneath the building slab. During March and April 2015, supplemental soil sampling was completed by a USEPA Superfund Technical Assessment and Response Team contractor who collected 86 surficial soil samples and 143 subsurface soil samples. Samples collected by the USEPA were analyzed for the constituents listed in appendix K, table K-02 by the USEPA Region 7 laboratory (Kansas City, Missouri) for inclusion in the BRA.

The former hat factory property was divided into areas A, B, C, and D based on site history and levels of PCE contamination in tree-core samples (fig. 2-07; appendix C, fig. C-01). Area A contains the former hat factory and land north of the factory. Apart from the concrete land cover of the former hat factory, the area is a mix of grass and wooded forest. Area A also contains the metal and central dumps. Area B is east of Area A, is entirely wooded forest, and is steeply sloped to the north. Area C is at the east part of the property and contains the eastern dump. Area D is south of Area A and is the location of a historical parking lot, which can be seen clearly in historical imagery (appendix A, figs. A-01 through A-03).

2.4.1 U.S. Geological Survey Missouri Water Science Center Soil Sampling

From January 2012 to September 2014, the USGS MO WSC characterized the nature and extent of

contamination in soils. Soil samples were collected from several excavated trenches on the former hat factory property. Direct-push methods were used to collected soil samples on the entire former hat factory property. Some split-spoon samples were collected using the cable-tool rig to validate refusal depths determined using direct-push methods.

Samples were collected according to methods described in the general project QAPP (USEPA, 2011b). In addition, a Field Sampling Plan (FSP; USEPA, 2014) was prepared describing each mobilization effort and sampling methods. Excavated trenches were sampled on several occasions to supplement direct-push efforts. In January 2012, the city of Vienna excavated a 54-in.-deep trench for a new water-distribution line along the west side of the former hat factory from Chestnut Street to 10th Street. This provided an opportunity to efficiently collect subsurface soil samples along the west side of the former hat factory property. A total of 37 samples ("SL" prefix on fig. 2-07) were collected from 1 to 3.5 ft deep in the trench about every 5 ft along the trench. On November 12, 2013, samples were collected from nine shallow (less than 3 ft deep) trenches dug to refusal in the eastern dump ("TR" prefix on fig. 3-03) near where several trees had detectable amounts of PCE and CFC-113 (appendix C, figs. C-01 and C-03). Seven trenches as much as 3 ft deep also were dug near the metal and central dumps (fig. 2-07). A total of 48 soil samples were collected from the seven trenches at 1-ft-deep intervals (sites VIN-TR-01 through VIN-TR-16; table B-02). On April 23, 2014, four samples were collected from 1 to 3 ft deep within an excavated trench during the replacement of water line at the northwest corning of the former hat factory (sites City-Trench-1 through City-Trench-4; table B-02). For all excavation samples, soil samples were collected from the wall of the excavation similar to how soil samples were collected from the soil cores collected by the Geoprobe[®].

Samples were collected from soil borings at boreholes GP-001A, GP-001B, GP-001C, GP-002A, GP-002B, GP-003A, and GP-004A and at GP-SHD-01 (fig. 2-07) during May, June, and November 2012 near where a former shed that was used to store chemicals (appendix A, figs. A-01 and A-02). Borehole depths ranged from 2 to 10 ft, but most were greater than 7 ft.

A total of 735 soil samples were collected from 154 soil borings by the USGS MO WSC during March and May 2014 as part of a site-wide screening effort. Of the 735 samples, 33 were laboratory samples submitted

to the USGS contract lab and analyzed for constituents on the HSL, 13 were sent to USGS contract laboratories and analyzed for only VOCs, 599 were analyzed for VOCs using the portable GC, and 90 were analyzed for total volatile organic compounds (TVOCs) using the portable GC. Sample collection was done in accordance with the Phase 3 FSP (USEPA, 2014). Of the 735 soil samples, 155 were surficial soil samples and were collected from 0 to 0.5 ft bgs (fig. 2-07). The remaining 580 soil samples were considered subsurface soil samples and were collected from 0.5 ft to refusal, which was typically about 5 ft but ranged from 1 to 18 ft deep. A total of 39 split soil samples from 11 locations were collected for analysis of VOCs by the USGS contract laboratory and the portable GC for verification of portable GC efficacy (appendix B, table B-02).

Figure 2-07. Locations of surficial, subsurface, and split-spoon soil sample locations collected from January 2012 to September 2014, Vienna, Missouri.

Between March and May 2014, sampling locations were selected based on an unbiased grid in the four areas (fig. 2-08) and a "walking-out" approach. A site-wide sampling grid was established, and locations were equally spaced within each area. Surficial soil samples were collected at all locations; however, subsurface soil samples were collected using a "walking-out" approach. Based on the tree-core reconnaissance sampling through April 2014, subsurface soil sampling originated near the tree or trees that had the largest PCE concentrations with the exception of area B, which had no PCE detections in tree-core samples. In area B, subsurface samples were collected from all grid locations accessible with the direct-push drill rig. For the "walking-out" approach, once an initial boring in each site area was done, the next four borings were collected in the four cardinal directions (N, S, E, and W) at the specified grid spacing, "walking-out" from the "hot spot" identified by the tree-core reconnaissance. Sampling then continued in each cardinal direction until the maximum PCE value at all depths in a boring was less than 55 micrograms per kilogram (µg/kg), or 1/10th of the previous residential preliminary remediation goal of 550 µg/kg, at which point the next grid point ($n+1$) in that direction was skipped and a boring was done at the second grid point ($n+2$) in that direction. If PCE did not exceed 55 µg/kg in samples from any depth in that boring ($n+2$), then no further sampling was done in that direction. When PCE in the ($n+2$) boring exceeded 55 µg/kg, then regular-grid boring resumed in the four cardinal directions from the ($n+2$) boring. If

borings exceeded the 55 µg/kg PCE level, indicating a restricted area of subsurface soil contamination, additional

borings were collected at about one-half the initial grid spacing to further refine the area of subsurface

contamination. Once screening of all borings in an area was completed, screening began in the next site area. A

second boring within 3 ft of the initial subsurface screening boring was drilled for the purpose of collecting

subsurface samples for laboratory analysis. Portable GC samples were also taken from the core at the same

interval where laboratory samples were collected to provide information on comparability of screening results

with laboratory results.

Of the 155 surficial soil samples, 109 were collected in March 2014 from 95 locations using a

composited-sample approach (fig. 2-07), and the remaining samples were collected from surface soils at

boreholes during the subsurface sampling in April and May. At each location, three equal-volume samples were

collected from a 2-ft radius and composited in a stainless steel bowl, and one sample was collected from the

homogenized sample. All the 155 surficial soil samples were analyzed by the portable GC for either TVOCs or

VOCs (appendix B, table B-02). The TVOC analysis was used because it quickly (less than 60 seconds) screens

for the presence of any VOCs. Of the 155 surficial samples, 15 were submitted to the USGS contract laboratory

for analysis of constituents on the HSL (fig. 2-04).

Subsurface samples were collected from March 2014 through May 2014 from 167 locations (fig. 2-07).

Of the 580 subsurface samples, 549 were analyzed for VOCs using the portable GC (appendix B, table B-02). A

total of 31 samples were submitted to the contract laboratory, and 13 samples were analyzed for VOCs only,

whereas 18 samples were analyzed for constituents on the HSL. Because of the "walking-out" approach used,

there were more subsurface soil sampling locations near the central dump and sycamore tree in area A that had

anomalously large concentrations of PCE, around the three trees south of the former hat factory, and below the

slotted concrete floor on the southwest corner of the former hat factory (fig. 2-07). Very few subsurface soil

samples were collected from area B because the area was heavily wooded and steeply sloped, preventing safe

access to sampling locations using the truck-mounted Geoprobe®. In addition, tree-core samples indicated PCE

concentrations were very low in area B (appendix C, fig. C-01).

Subsurface soil samples were collected below the concrete building slab (subslab samples) from more

than 50 locations (fig. 2-07). Subslab locations were selected based on building operational history and the presence of specific features such as floor drains or suspected solvent-use areas (fig. 1-09). Once the concrete slab was breached using a hammer bit, a continuous core was collected. In total, two HSL samples and three laboratory VOC samples (four subsurface and one split-spoon sample) were collected from below the slab (fig. 2-04). Additionally, one HSL sample and two laboratory VOC samples were collected from below the slab in the basement.

A background surficial and subsurface HSL sample from undeveloped pasture land was collected (SBG-01; fig. 2-04). The location of the background sample was selected based on property access but also because it is near the former hat factory property yet topographically upslope or across the drainage basin at a similar elevation in an attempt to ensure similar soil characteristics.

Surficial and subsurface sampling was done along the building perimeter and in loading dock areas or exterior doors where spills or intentional dumping of wastes may have occurred. Locations were sampled about every 100 ft around the perimeter of the building (about 1,000 linear ft) with locations adjusted to ensure at least one location was sampled adjacent to each exterior door. A total of seven laboratory samples (one surficial HSL [site P010], three subsurface HSL [sites P006, D063, and D069], and three laboratory VOC [sites D066, D068, and D019]) were collected along the building perimeter (fig. 2-04).

Samples were collected from all three identified dump sites within the property (metal, central, and east dumps; figs. 1-08 and 2-07). Surficial and subsurface screening samples were collected from each dump site with the exception of the metal dump where a subsurface sample was not collected because of safety concerns. A HSL sample was collected from each dump area (sites AM01, A018, and C003; fig. 2-04).

During August and September 2014, split-spoon sampling was completed across the former hat factory property using a cable-tool drill rig. Split-spoon sampling was completed to validate refusal depths determined by the truck-mounted Geoprobe®. Samples were collected by drilling to a certain depth with the cable-tool rig then driving a 3-in., split-spoon sampler. In total, 39 soil samples were collected from 11 split-spoon locations, including 4 laboratory samples analyzed for only VOCs and 35 portable GC samples (figs. 2-04 and 2-07; appendix B, table B-02).

2.4.2 U.S. Environmental Protection Agency Soil Sampling

Because of concerns from the USEPA that an insufficient number of laboratory samples were collected for the BRA, an additional 229 soil samples were collected by the USEPA and Tetra Tech, Inc., from March 23 to April 6, 2015 (appendix B, table B-10). The USEPA subdivided USGS-defined areas A, B, C, and D into exposure units based on historical use, geography, vegetative cover, and levels of contamination in historical samples (Tetra Tech, Inc., 2015). Surficial soil samples were used to evaluate risk in all areas assuming the threat of contamination for residential land. Subsurface samples were used to evaluate risk in the respective areas based on exposure to construction workers during excavation activities. In contrast to the 0 to 6 in. designated by the USGS, surficial soils collected the USEPA were from 0 to 2 in. bgs using stainless steel tablespoons. Subsurface (greater than 2 in.) samples were collected using a track-mounted Geoprobe® drill rig. All samples were analyzed by the USEPA Region 7 laboratory in Kansas City, Missouri.

In total, 110, 37, 45, and 37 soil sample locations were established within area A, B, C and D, respectively (appendix B, table B-10). In area A, surficial soil samples were collected from all cells off the building foundation and subsurface soil samples were collected from each cell except for cells A-4 to A-6 (fig. 2-08). In areas B and C, surficial soil samples were collected in all cells, but subsurface soil samples were collected from only 9 out of 16 cells in area B and 13 out of 16 cells in area C because of shallow refusal or inaccessibility of the Geoprobe® rig to the cell. In area D, surficial and subsurface soil samples were collected from 17 out of 21 cells because of overlying concrete or shallow refusals.

Figure 2-08. Locations of surficial and subsurface soil sample locations collected by the U.S. Environmental Protection Agency, Vienna, Missouri, 2015.

2.5 Sanitary Sewer and Septic Investigation

Sampling of the city of Vienna municipal sanitary sewer, as well as a former on-site septic system on the former hat factory property, was completed during two major events during January through March 2012 and May 2014 (fig. 2-09). Sampling of the sanitary sewer and on-site septic system was completed in accordance with

the QAPP (USEPA, 2011b) and FSP (USEPA, 2014), respectively. A total of 52 effluent samples were collected

from 38 locations in the city sanitary sewer system and analyzed for PCE and other VOCs using the portable GC

(appendix B, table B-01).

Reconnaissance sanitary sewer sampling took place from January 30 to 31, 2012, with a screening of

selected sewer manhole junction boxes throughout the city to identify possible contamination sources. Effluent

from the former hat factory likely flowed from site MH-068 through a series of gravity sanitary sewer lines to the

lift station at site MH-043 where it was transported to the wastewater treatment plant on the west side of town

(fig. 2-09). Sanitary sewer sampling on February 22, 2012, and March 12, 2012, was focused on areas where

January reconnaissance samples indicated PCE detections.

Figure 2-09. Locations of field gas chromatograph (GC) and laboratory volatile organic compound (VOC) sewer

samples, Vienna, Missouri, 2012.

During site-clearing activities and trench sampling on November 2013, a septic tank was uncovered about

60 ft northwest of the northwest corner of the east extension (site Septic1; fig. 2-07). Attempts to trace the clay

tile line exiting the septic tank were unsuccessful because the pipe was clogged with soil. Several other drain lines

were located using the utility locator including the slotted-floor drain, two bathroom drains on the south side of

the building, a bathroom drain emanating from the basement, and one drain emanating from the southwest corner

of the basement (fig. 2-07). The slotted-floor drain was traced from the west part of the slotted floor about 12 ft

west to what is likely an old culvert and then north about 8 ft where it terminated about 20 ft short of a tinhorn

effluent pipe. One bathroom drain was traced from a PVC cleanout about 90 ft west to the location of the

bathrooms where it terminated. Another cleanout on the southeast corner of the building was traced about 120 ft

to the northeast where it terminated. Upon excavation of the ground directly south of the bathrooms, two clay

tiles leaving the bathrooms less than 1 ft bgs were located. Further excavation near the southeast corner of the

building revealed that the same two drains less than 1 ft bgs ran in parallel to the corner of the building then split

off, with the 6-in. drain turning north towards the previously excavated septic tank and the 4-in. drain continuing

northeast about 120 ft. Excavation north of the septic tank revealed tile line leading to a large (30 ft long x 50 ft

wide) sand filter. A screening sample was collected in this sand filter (borehole A-SAND; fig. 2-07). An additional cleanout south of the north fence line was traced southeast about 50 ft toward the termination of the 4-in. drain and about 165 ft to the north in the direction of site MH-067 (fig. 2-09). On May 7, 2014, two drain lines in the basement of the north extension were traced. The first drain was connected to the lavatories on the east wall of the basement and was traced about 5 ft east then 70 ft north along the building towards site MH-068 (fig. 2-09). The second drain was in the southwest corner of the basement on a 2-ft, raised concrete slab. The cable was fed about 19 ft, but attempts to trace the line below the concrete platform outside were unsuccessful, possibly because of rebar in the concrete. Three subsurface HSL samples (boreholes A064, A028, and A053) were collected in the vicinity of the 4-in. drain lines between the two terminations, and one HSL sample (borehole D063) was collected near the PVC cleanout of the 4- and 6-in. bathroom drains (fig. 2-04).

2.6 Vapor-Intrusion Sampling

The USEPA collected vapor-intrusion samples from 16 to 17 residences in the vicinity of the former hat factory property between April 2014 and January 2015 (appendix J, figs. J-01 through J-03; table J-01) in accordance with the approved QAPP (USEPA, 2017). Indoor surveys were completed at properties where sampling data indicated a possible indoor air source. Site-wide ambient air samples were not collected, but each property was sampled by collocating indoor air with subslab/crawlspace samples. Time-integrated samples were collected during a 24-hour period using Summa™ canisters. Typically, two or more Summa™ canisters were deployed inside each home with one targeting below the foundation (either subslab or crawlspace samples) and a second targeting indoor living areas such as a living room or bedroom. Two field duplicates (one indoor-air and one subslab sample) were collected from one home during each sampling round. After deployment, the canisters were collected by the USEPA and transported to the USEPA Region 7 laboratory in Kansas City, Kansas, for analysis. Multiple lines of evidence described in the vapor-intrusion summary report were evaluated for samples that exceeded the USEPA Region 7 action level of 2.0 micrograms per cubic meter ($\mu g/m^3$) for TCE to determine whether the vapor intrusion pathway was complete (USEPA, 2017).

In April 2014, 36 vapor-intrusion samples were collected from 16 residences. Indoor-air samples were

collected at all locations; however, a subslab or crawlspace sample was not collected at one location (site VIN-VI-15) (table J-1). Additionally, a grab sample was collected from air inside of the cistern at site VIN-VI-01 (JW-01). In July 2014, 37 vapor-intrusion samples were collected from 17 residences and included the addition of 1 location (site VIN-VI-17) and a field blank. Indoor-air samples were collected at all locations; however, subslab or crawlspace samples were not collected from three locations (sites VIN-VI-15, VIN-VI-7, and VIN-VI-11). In October 2014, 38 vapor-intrusion samples (including a field blank) were collected from 17 residences. Indoor-air samples were collected at all locations; however, no subslab or crawlspace samples were collected from two locations (sites VIN-VI-15 and VIN-VI-7). In January 2015, 36 vapor-intrusion samples (including a field blank) were collected from 16 residences. Indoor-air samples were collected at all locations; however, no subslab or crawlspace samples were collected from two locations (sites VIN-VI-15 and VIN-VI-7).

2.7 City Water-Supply Distribution System

To verify the effectiveness of the treatment system at well COV-03, water-distribution samples were collected during a reconnaissance sampling of the Vienna public-water supply distribution system on June 12, 2014. An attempt was made to collect samples at the locations sampled during the SI (fig. 1-07); however, a sample was not collected at WOW Pizza, a location at the end of the distribution system, because that business had closed since the sampling in 2007. A replacement end-of-distribution location was sampled at the South Central Regional Stockyard, Inc., from a cleaning room closet (site VIN-STOCKYARD; fig. 1-02; appendix B, table B-01). The other location sampled during the SI (Vienna City Hall, site VIN-CITYHALL) was sampled in addition to a third location at the Vienna Ball Park (site VIN-BALLPARK). Laboratory VOC samples were collected from taps after a 5-minute purge.

2.8 Analytical Methods Used in the Remedial Investigation

2.8.1 Soil and Water

Although the BRA assessed risk from VOCs, SVOCs, organochlorine pesticides, polychlorinated biphenyls, organophosphorus pesticides, inorganic anions, and metals, the target compounds for the Vienna Wells

RI were PCE and its degradation products (TCE and *cis*-DCE). The USGS used a portable GC to rapidly and inexpensively determine concentrations of PCE and other VOCs including TCE, *cis*-DCE, *trans*-1,2-dichloroethene, benzene, toluene, and CFC-113 in soil and water samples. The portable GC also was used to analyze samples of drill cuttings during monitoring well installation to guide well design and determine the disposition of drill cuttings. The portable GC also was used extensively during the reconnaissance sampling of surface water and springs. Split samples from each of these media were submitted to the analytical laboratory for verification of the portable GC results. Although a large number of soil samples were collected from soil cores (generally one sample every 2 to 3 ft in depth) and analyzed by the portable GC to develop a detailed understanding of VOC contamination in the subsurface, only laboratory analytical results were used in the BRA calculations.

The method detection limit (MDL) for the portable GC (table 2-03) was determined using USEPA methods (Wisconsin Department of Natural Resources, 1996). Because the portable GC is used as a screening tool, concentrations below the MDL were not censored; however, concentrations below the MDL were not used when comparing portable GC data with laboratory data. Hydrocarbons from an oil-core inside the cable on the cable-tool drill rig resulted in numerous small hydrocarbon peaks that interfered with analysis of *cis*-DCE, toluene, and benzene by the portable GC, and detections of benzene and toluene in drill-cutting samples are likely attributed to drilling activities.

Table 2-03. Method detection limits for constituents analyzed using the portable gas chromatograph.

A comprehensive discussion of the quality and comparability of the portable GC data to established laboratory methods was presented in the New Haven OU1 and OU3 RI (USEPA, 2003). Identical methods and equipment used in the OU1 and OU3 RI were used for the Vienna Wells RI. In addition to required daily check standards, duplicates, and blanks (appendix B, table B-04) as a check for continued reliability of the portable GC performance, split samples from soils and groundwater sites were analyzed by the portable GC and the USGS contract laboratories. The concentrations spanned several orders of magnitude; therefore, the data were base 10 log-transformed to calculate correlation coefficients. Pearson's coefficient of determination (R^2) determined by

comparing results of soil samples with PCE concentrations above the MDL analyzed by the portable GC and the analytical laboratory indicated that the portable GC provided reliable estimates of PCE (n=16; R^2=0.89) concentrations in soils and plotted reasonably close to, although slightly rotated clockwise from, the equal value line indicating equivalence between laboratory-determined and portable GC-determined concentrations (fig. 2-10). The rotation of the soil regression line about 100 µg/L on the equal value line indicates a small underestimation above 100 µg/L and small overestimation below 100 µg/L. In a similar comparison of water samples with VOC detections analyzed by the portable GC and the analytical laboratory, the correlation coefficient for PCE (n=59; R^2=0.98) indicated that the portable GC results provided reliable estimates of PCE concentrations in water samples and plotted very close to the equal value line (fig. 2-10). Other VOCs (TCE, *cis*-DCE, benzene, and toluene) were not detected enough in portable GC samples to develop correlations between portable GC and analytical laboratory results.

Figure 2-10. Relation between concentrations of tetrachloroethene (PCE) in laboratory samples and portable gas chromatograph (GC) samples in soil and water samples collected at the Vienna Wells site, Vienna, Missouri, 2011-2016.

Detailed information about the laboratory analytical methods and their quality-assurance procedures are described in the RI QAPP (USEPA, 2011b). Generally, soil and water samples were analyzed at one of the USGS contract laboratories for PCE and a variety of other constituents. See Appendixes K and B for summaries of laboratory analytes (appendix K, tables K-01 and K-02) and samples submitted for laboratory analyses (appendix B, tables B-06 and B-07).

Quality-assurance laboratory samples were collected by the USGS and USEPA during the Vienna Wells site RI. These samples consisted of duplicate samples (29) and trip blanks (12). Only 2 of the 19 USEPA soil sample duplicate pairs contained PCE detections. The relative percent difference (RPD) was calculated by divided the difference between the concentration values of the sample pair by the average concentration of the sample pair values. In one set the RPD was 9.8 percent, and the other sample pair had a sample that did not have

a PCE detection above the reporting limit of 2.80 µg/kg, but the corresponding duplicate sample had 6.4 µg/kg of PCE, resulting in a RPD of 78 when using the detection limit for the environmental sample. Neither of the two USGS laboratory soil sample duplicate pairs contained PCE detections or detections of other VOCs (TCE, *cis*-DCE, benzene, and toluene). The overall average RPD for soil based on the detection limits of these two sample sets was 44. The average RPD between the eight duplicate pairs for water samples was substantially lower (3.1) for PCE. There were no detections of PCE in any of the trip blanks, indicating that the decontamination and sample handling protocols followed in the field were adequate to prevent cross-contamination of environmental samples and met the quality-assurance goals in the project QAPP.

2.8.2 Tree Cores

Tree-core samples were analyzed by the Missouri University of Science and Technology Environmental Research Center in Rolla, Missouri, for analysis by GC using established methods (Limmer and others, 2014). Although CFC-113 was detected in several tree-core samples, calibration and determination of a MDL was difficult because of its volatility, so an MDL was not determined. All concentrations in tree-core samples were reported as mass of constituent per volume of water in the tree core.

All trip and field blanks were below the reporting limit for PCE, TCE, *cis*-DCE, and CFC-113 except for the samples collected on October 29, 2015 (fig. 2-01), when two field blanks collected during the sampling had 150 and 121 nanograms per liter (ng/L) of TCE. Trichloroethene is sometimes present in supplies used in tree coring such as gloves; therefore, contamination from field supplies is likely the cause of the high occurrence of TCE detections to the southeast of the former hat factory property (appendix C, fig. C-02).

3.0 Nature and Extent of Contamination at the Vienna Wells Site

The following sections focus on the nature and extent of constituents in vegetation, surface water and springs, soils, and groundwater at the Vienna Wells site identified as contaminants of concern (COCs; table 3-01) by the BRA or that were used at the former hat factory. Other constituents that were identified as contaminants of potential concern (COPCs; table 3-01) include the COCs, as well as other constituents that did not constitute a

substantive risk, were included in the baseline risk assessment but were not included in the discussion in the following sections. A summary of reconnaissance sampling results of the city sanitary sewer system and indoor-air sampling also is presented, followed by a summary of the primary contaminant source areas investigated.

Table 3-01. List of contaminants of concern and contaminants of potential concern at the Vienna Wells site, Vienna, Missouri.

3.1 Distribution of Tetrachloroethene and Other Volatile Organic Compounds in Tree-Core Samples

Of the 488 trees sampled and analyzed for PCE, TCE, *cis*-DCE, and CFC-113, 324 and 56 had detections greater than the MDL for PCE and TCE, respectively (appendix C, figs. C-01 and C-02; appendix B, table B-05). When looking at the maximum concentration measured from each tree, 230 and 16 trees had concentrations above the reporting limit of 4.7 and 77 ng/L for PCE and TCE, respectively. Also, 46 trees had detections of CFC-113 (fig. C-03). Because of the large volatility of CFC-113, detectable CFC-113 concentrations in tree-core samples suggest substantial CFC-113 concentrations in the subsurface. Trees with large PCE concentrations (greater than 1,000 ng/L) were clustered near the former hat factory to the north and south, near the central dump, and near the eastern property line (fig. C-01). The largest PCE concentrations were measured in a sycamore tree near the central dump at 11,191 ng/L (site TC032) and in trees directly south of the slotted floors at concentrations of 5,337; 10,582; 5,889; and 9,236 ng/L (sites TC014, TC013, TC012, and TC020, respectively). No tree-core sample contained detectable concentrations of *cis*-DCE.

Large PCE concentrations in trees northeast of the former hat factory coincide with the location of the central dump and may be the result of discarded PCE-containing materials because of the large amount of debris present. These concentrations could also be the result of contamination from a potentially broken sewer line identified during the delineation of on-site drains (fig. 3-01) or from migrating groundwater from the vicinity of the slotted floor because some smaller diameter trees along the north perimeter of the building had detections and no larger diameter trees were present between the former hat factory and the cluster northeast of the former hat

factory (appendix C, fig. C-01). Large PCE concentrations in trees south of the former hat factory indicate subsurface contamination extending to the south property line. Moderate PCE concentrations in trees near the eastern property boundary indicate another area of shallow subsurface contamination (fig. C-01). Concentrations of TCE in trees generally were large in areas where concentrations of PCE were large (fig. C-02), except to the southeast. Large TCE concentrations to the southeast may be indicative of shallow transport of TCE but are more likely the results of sample contamination (see section 2.8.2, "Tree Cores").

3.2 Distribution of Tetrachloroethene in Surface Water and Springs

Only two springs contained detectable concentrations of PCE (appendix D, fig. D-09; appendix B, table B-01). The spring at site WP-020 contained PCE concentrations ranging from below detection to 13 µg/L during sampling in 2012, 2013, 2014, 2015, and 2016. The second spring location with PCE detections was site JW-Seep, which had an estimated 2.6 µg/L of PCE during April 2014. Subsequent lab samples collected from site JW-Seep contained concentrations of PCE ranging from below detection to an estimated 0.96 µg/L.

Only one stream sample location (site TB2-Trickle) had a small PCE concentration (2.2 µg/L on March 19, 2014; appendix D, fig. D-09); however, all subsequent samples (sites TB2-2A, TB2-2B, TB2-2C, and TB2-2D) collected on April 17, 2014, and May 19, 2014, did not contain detectable PCE concentrations. Because the only detection of PCE at site TB2-Trickle was collected from a small input into the East Tributary after recent (less than 24 hours) rainfall, PCE does not seem to be present in surface water under base flow conditions.

3.3 Distribution of Tetrachloroethene and Other Constituents in Soils and Sediment at the Vienna Wells Site

Of the 1,185 soil samples collected by the USGS and USEPA for laboratory and portable GC analysis, PCE was detected in 25 surficial soil samples and 284 subsurface soil samples (appendix B, tables B-02, B-06, and B-10; figs. 3-01 through 3-04; appendix I, figs. I-10 through I-19); however, PCE was detected in only two soil samples above the USEPA Region 7 residential screening level for soil of 24,000 µg/kg.

Figure 3-01. Maximum concentration of tetrachloroethene (PCE) in soil samples in each soil boring collected by

the U.S. Environmental Protection Agency (USEPA) and U.S. Geological Survey (USGS) at the Vienna Wells site, Vienna, Missouri.

All soil samples collected from trenches and excavations extending to more than 3 ft in depth did not have detectable PCE concentrations. Concentrations of PCE were largest in soil samples in two of the three areas of contamination indicated by tree-core sampling: under and in the immediate vicinity of the former hat factory and northeast of the former hat factory near the central dump (appendix C, fig. C-01).

At and in the immediate vicinity of the former hat factory, concentrations of PCE were largest below the concrete slab near the southwest corner where the soil is less than 6 ft thick. PCE concentrations ranged from less than detection to 51,840 µg/kg at depths of 0.6 to 12.3 ft bgs in the 93 samples collected from boreholes beneath the slotted floor and beneath the solid slab south of the slotted floor. The two largest PCE concentrations were below the concrete slab, 2 ft bgs at borehole D024 (26,424 µg/kg, fig. 3-04), and 5 ft bgs at borehole D030 (51,840 µg/kg); and both boreholes gave off sweet odors. Because of the concurrently large PCE concentrations and sweet odors associated with pure PCE product, it is likely that there is residual nonaqueous phase liquid below the slotted floor. Below the concrete slab, the depth of PCE contamination typically was less than 5 ft (appendix I, figs. I-10 through I-19). In borehole D024, concentrations decreased from 26,424 µg/kg at 2 ft bgs to 240 µg/kg at 3.3 ft bgs. In contrast, concentrations increased in borehole D030 with depth and were 3,023 µg/kg at 2 ft bgs, 4,800 µg/kg at 4 ft bgs, and 51,840 µg/kg at 6 ft. Areas where PCE concentrations decrease with depth, such as borehole D024, indicate probable contaminant source zones as concentrations spread out and are transported vertically. In contrast, areas where PCE concentrations increase with depth, such as borehole D030, indicate zones adjacent to contaminant source zones. Concentrations of PCE in drill-cutting samples collected during drilling of two nearby monitoring wells (wells MW-06 and MW-07) decreased with depth. In well MW-07, PCE concentrations in drill-cutting samples decreased from 1,920 µg/kg at 0 to 5 ft bgs to 82.8 µg/kg at 21 ft bgs at bedrock and to 2.10 µg/kg at 53 ft bgs at the bottom of the well.

Figure 3-02. Maximum concentration of tetrachloroethene (PCE) in selected laboratory soil samples collected by

the U.S. Environmental Protection Agency (USEPA) and U.S. Geological Survey (USGS) at the Vienna Wells site, Vienna, Missouri.

Figure 3-03. Maximum concentration of tetrachloroethene (PCE) in soil samples in each soil boring collected by the U.S. Geological Survey (USGS) and analyzed with a portable gas chromatograph at the Vienna Wells site, Vienna, Missouri.

Figure 3-04. Maximum concentration of tetrachloroethene (PCE) in select soil samples collected by the U.S. Geological Survey (USGS) and analyzed with a portable gas chromatograph at the Vienna Wells site, Vienna, Missouri.

Soil samples collected directly south of the former hat factory and extending to the southern property line had PCE concentrations ranging from less than detection to 1,968 µg/kg in samples collected from 0 ft to 10.9 ft bgs (figs. 3-01 and 3-02; appendix B, table B-02). The largest PCE concentrations were closer to the former hat factory and decreased toward the southern property line. Concentrations of PCE were largest in samples from between 5 and 9 ft bgs, corresponding in elevation to a depth of 0 to 4 ft below the former hat factory slab.

Soil samples collected along the southern edge of the former hat factory (boreholes SB-01 through SB-14) parallel to the septic lines leading to the septic tank north of the former hat factory and the city sewer manhole site MH-067 typically contained detectable concentrations of PCE. Concentrations of PCE in the 45 samples collected near the septic and sewer lines ranged from below detection to 3,892 µg/kg at depths ranging from 0.2 to 12.3 ft bgs, and decreased with distance to the east at a depth below 3.5 ft bgs. Because these PCE detections are at depths greater than 3 ft bgs, 2 ft deeper than the depth of the septic and sewer lines, and the presence of small pockets of gravel-sized chert and clayey sand underlain by beds of chert that could allow for horizontal flow of contamination are a few feet bgs south and underneath the former hat factory, it is likely that these detections are an extension of the plume under the slab at depth rather than from the sewer or septic lines themselves.

Concentrations of PCE under the basement slab, although above detection, were relatively low compared

to other soil samples. A total of 45 soil samples (42 portable GC and 3 USGS lab samples) were collected from the basement in the north extension of the former hat factory. Concentrations of PCE under the basement slab ranged from below detection to 379 µg/kg, and sample depths ranged from 0.5 to 9.6 ft bgs. The largest PCE concentrations in the basement were clustered in the southwest corner in boreholes A056, A057, A060, A061, A062, and A063 (fig. 3-04). Although concentrations were low, they increased slightly with depth. Because the depth of the basement floor is about 4 ft bgs at the level of the trench west of the former hat factory, the depth of these samples in relation to the samples from the trench are between about 7 to 14 ft bgs. Because the PCE concentrations in these samples are low and tree-core data indicate no PCE contamination west of Chestnut Street, contamination likely does not extend much west of the basement; however, soil sampling to depths around 15 ft west of the basement would be necessary to validate the extent of contamination.

Northeast of the former hat factory near the central dump (figs. 3-03 and 3-04), PCE concentrations ranged from less than detection to 15,552 µg/kg in portable GC samples, and sample depths ranged from 0 ft to 14.6 ft (appendix B, table B-02). Most PCE detections were between 4 and 11.5 ft, and the largest PCE concentrations were in borehole A028 at 10 (15,552 µg/kg) and 11.2 ft (9,588 µg/kg). Concentrations of PCE generally increased with depth in this area and were all below detection until 4 ft bgs, and the five samples that were above 100 µg/kg were at or below 6 ft bgs, suggesting contamination may have originated from local dumping at depth. One laboratory sample, collected in borehole A018 between 5 and 6.5 ft bgs, contained an estimated naphthalene concentration of 180 µg/kg (appendix B, table B-06). Because naphthalene is commonly associated with moth balls, and former employee accounts indicate hides were stored with mothballs, it is likely that spent hides and mothballs were discarded in the central dump. Although the largest PCE concentrations in the central dump are more than 30 ft lower in elevation from the largest PCE concentrations under the slab of the former hat factory, it is unlikely that PCE migrated through the soil from under the slab because of the large distance and similarity in magnitude of PCE concentrations. Additionally, all the boreholes between the south part of the former hat factory and the central dump do not contain large PCE concentrations, although land surface slopes heavily downward toward the central dump. It is possible, however unlikely, that PCE could have migrated from the slab to the central dump through preferential pathways. Another potential origin of PCE may

be from PCE disposal in the sewer line, which is discussed in section 3.5, and likely collapsed near the central dump. Additional soil sampling from and in the vicinity of the central dump and between the central dump and the former hat factory to elevations comparable to samples collected from and in the vicinity of the central dump may be necessary to determine the source of PCE contamination in the central dump.

Results from laboratory and portable GC analyses of soils samples were used with the geographical information system (GIS) program ArcMap™ (Esri, Redlands, California) to make interpolated surfaces of estimated PCE concentrations in soil at several depths bgs and to estimate the volume of contaminated soil and total mass of PCE (table 3-02). Interpolated surface were made at between 0 and 0.5 ft (appendix I, fig. I-10), 0.5 and 2 ft (fig. I-11), and in 2-ft intervals from 2 to 16 ft (figs. I-12 through I-19). Surfaces were interpolated using an inverse-distance-weighted routine on a 1-meter (m) grid. From this interpolation, it was estimated that 1,500,000 cubic feet or 56,000 cubic yards of PCE-contaminated soils are above the protection-to-groundwater threshold of 5.1 µg/kg with about 700,000 cubic feet or 26,000 cubic yards of PCE-contaminated soils above about 100 µg/kg (figs. I-10 through I-19). These soils contain a total estimated 33 kilograms (kg) (about 5.5 gallons of pure product) of PCE. The total mass of PCE was calculated by multiplying the average concentration within each concentration interval at each depth interval by the interval thickness and the area of that concentration interval. A cautionary note on the estimates; interpolation of concentration data by the GIS algorithm from a small number of points can produce features in areas without data points that are influenced by large differences among nearby data values, which may not depict the actual extent of concentrations.

Table 3-02. Extent and magnitude of tetrachloroethylene (PCE) contamination at various depth intervals below ground surface (bgs), Vienna Wells site, Vienna, Missouri.

The contaminated area indicated by tree-core sampling on the east part of the former hat factory property contained less than 30 µg/kg of PCE in surficial soil samples collected from 3 (C001, C002, and C003) out of more than 100 soil sampling sites. Sample depths ranged from 0 to 8 ft bgs; however, depths of refusal were shallow because of the presence of shallow sandstone bedrock, and most samples were collected at depths less than 4 ft.

This discrepancy between tree-core and soil data exists because many trees were sampled across the property line, and soil sampling across the property boundary was not plausible because of steep slopes and heavily-wooded areas. Groundwater is about 45 ft bgs in the east part of the site, and it is plausible that tree roots have grown to comparable depths. The discrepancy likely exists because PCE concentrations in tree-core samples are a result of PCE vapors and groundwater transport into trees below the restrictive sandstone bedrock from the same source as the plume from the former hat factory.

Large concentrations of TCE, *cis*-DCE, and CFC-113 were generally in areas with large PCE concentrations. Trichloroethene was detected by the portable GC in 37 soil samples either in the immediate vicinity of the former hat factory or to northeast of the former hat factory near the central dump (fig. I-01). In these areas, concentrations of TCE ranged from below detection to 4,256 µg/kg in borehole A028 at depths ranging from 0 to 12 ft bgs. Similar to TCE, *cis*-DCE was detected by the portable GC in 16 soil samples either in the immediate vicinity of the former hat factory or northeast of the former hat factory near the central dump (appendix I, fig. I-02). In these areas, concentrations of *cis*-DCE ranged from below detection to 246 µg/kg in borehole D030 from depths ranging from 0.9 to 11.2 ft. Because TCE and *cis*-DCE were not likely used during the hat-making process and are only present in samples with large PCE concentrations, the presence of TCE and *cis*-DCE are either the result of PCE degradation or were contaminants in the original PCE solvent because manufactured PCE solvent can contain percentage-range concentrations of TCE and *cis*-DCE (Morrison and Murphy, 2013). Concentrations of CFC-113 were largest under the slotted floor on the south side of the former hat factory, under and in the vicinity of the basement of the former hat factory, and in a streambed sediment sample (site TB1-2; fig. I-03). The presence of CFC-113 in areas of large PCE concentrations indicates CFC-113 was likely dumped similarly to PCE and may have been used in place of PCE.

None of the three streambed sediment samples had detectable concentrations of any COC. Some COPCs (table 3-01) were detected at low levels and did not constitute a substantive risk but are included in the baseline ERA (appendix B, table B-06; appendix L, tables L-02 and L-04).

Several COCs in surficial soil samples above concentrations in the background sample from borehole SBG-01 were identified in the ERA including the polycyclic-aromatic hydrocarbons (PAHs) benzo[*a*]anthracene,

benzo[*a*]pyrene, benzo[*b*]fluoranthene, dibenzo[*ah*]anthracene, and Ideno[1,2,3-*cd*]pyrene (appendix I, fig. I-04); and metals aluminum, arsenic, chromium, manganese, and thallium (appendix I, figs. I-05 through I-09). These metal contaminants are naturally present and do not innately indicate site-derived contamination. Figures summarizing the spatial distribution for each of these constituents are in appendix I (figs. I-05 through I-09) and appendix B (tables B-02, B-06, and B-10). Most PAH and metal detections were in the vicinity of the former hat factory and near the metal and central dumps with the exception of a large chromium concentration in a sample (site C11) from the east dump (fig. I-07) and is presumably related to the piles of old fur and leather strips, which are known to contain chromium when tanned (Bienkiewicz, 1983), in the vicinity of the sample. The large concentrations of PAHs and metals in the metal and central dumps corroborates field observations of roofing material, large metal debris, old drums, bricks, and various metal cans in these dumps.

3.4 Groundwater Flow and Nature and Extent of Volatile Organic Compound Contamination in Groundwater at the Vienna Wells Site

3.4.1 Groundwater Flow

The potentiometric surface in the Vienna area was refined from the regional scale using data from domestic wells and surface-water features in the vicinity of Vienna (figs. 2-05). This local potentiometric map indicates a recharge area southwest of Vienna and a groundwater divide extending roughly along U.S. Highway 63 in the city. Groundwater west of this divide flows toward the Maries River and Fly Creek. East of the divide, groundwater flows east to the Gasconade River. Hydraulic gradients west of the divide are generally uniform and low (less than 50 ft/mi). In contrast, gradients to the east generally are steeper (21 to 3,000 ft/mi). Pumping of public-supply well COV-03 induces a depression in the water table of about 80 ft near the well and likely captures any deep (less than about 710 ft elevation) groundwater in the vicinity of the former hat factory property (figs. 2-06 and 3-06). Although no water-level data exist to characterize the potential cone of depression caused by pumping in public-supply well COV-01 (fig. 3-05) before it was put on standby in 2007, it is likely that its effects were comparable to those of public-supply well COV-03 (fig. 3-06) and also captured deep groundwater

in the vicinity of the former hat factory property. Shallow (greater than about 710 ft elevation) groundwater and

groundwater far from the former hat factory property, such as groundwater in monitoring wells MW-05S, MW-05D, MW-09S, and MW-09D, likely is not captured by the pumping of public-supply well COV-03.

Figure 3-05. Generalized geohydrologic section *A-A′* in and near Vienna, Missouri.

Figure 3-06. Generalized geohydrologic section *B-B′* in and near Vienna, Missouri.

A more detailed potentiometric map at the former hat factory property was constructed from water levels

collected during March 2016 from shallow monitoring wells (appendix D, fig. D-01). This potentiometric map,

which did not show substantial seasonal variation, indicates the shallow local flow system is dominated by local

topography and likely is bounded by the East Tributary, which acts as a drain around an elevation of 700 to 710 ft

(fig. 2-05). Continuous water levels measured in monitoring wells MW-06, MW-06A, and MW-07 also indicated

that the pumping of public-supply well COV-03 has a substantial effect on the shallow local flow system at the

Vienna Wells site as water levels in these wells decrease about 0.5 ft soon after the pump in public-supply well

COV-03 is turned on (for example, see fig. D-11). Local groundwater at the former hat factory property flows

primarily to the northeast on the west side of the property, likely because of the influence of pumping from

public-supply well COV-03, and transitions more eastward in the area east of the site where it emerges as a spring

at site WP-020 near the East Tributary. Horizontal hydraulic gradients vary between about 0.15 foot per foot

(ft/ft) in the vicinity of the former hat factory to about 0.05 ft/ft east of the former hat factory. Water-level data

collected from monitoring well clusters MW-01, MW-04, MW-05, and MW-06 indicated a steep vertical gradient

of 0.51, 0.58, 0.34, and 0.53 ft/ft (average=0.49 ft/ft), respectively (table 3-03). Although all monitoring wells at

the Vienna Wells site are within a 0.5-kilometer area, this steep vertical gradient indicates that the horizontal K

value in these areas may be much larger than the vertical K value because there is a large amount of vertical

resistance to head. A ratio of horizontal to vertical K values of 10:1 generally is used in modeling groundwater

flow in the Ozark aquifer (Imes and Emmet, 1994). In contrast, water-level data collected from monitoring well

clusters MW-03 directly north of the former hat factory, MW-08 northeast of the former hat factory, and MW-09

adjacent to the East Tributary indicated a small vertical hydraulic gradient of 0.09, 0.02, and 0.05 ft/ft, respectively. This small gradient indicates that the vertical K value in the vicinity of these wells may be similar to the horizontal K value. Small vertical gradients at the Vienna Wells site are likely caused by the presence of fractures and more permeable material. Likewise, large vertical gradients are likely caused by the lack of fractures and less permeable material.

Table 3-03. Average calculated vertical hydraulic gradients in monitoring well clusters, Vienna Wells site, Vienna, Missouri.

Slug tests completed as part of this RI indicated horizontal K value estimates ranged from 0.04 to 50 ft/d in wells open to the Roubidoux Formation and 0.7 ft/d in well MW-08 in the Eminence Dolomite (appendix F, table F-1). Well MW-01 had the smallest K values (0.04 and 0.3 ft/d), which are about one to two orders of magnitude smaller than the other monitoring wells. Well MW-09D had the largest K values, which ranged from 20 to 50 ft/d, and likely was installed in a large amount of fractured rock because of its proximity to a nearby stream. The remaining wells have K values ranging from 0.8 to 7 ft/d.

The rate of groundwater flow is a function of the hydraulic gradient, aquifer porosity, and K value of the aquifer. The rate of advective movement (velocity) of groundwater through an aquifer is described by Darcy's Law:

$$v_a = \frac{K\frac{dH}{dX}}{n_e}$$

where

- v_a is the average linear velocity, in feet per day;

- K is the hydraulic conductivity of the aquifer (measure of the ability of a porous media to transmit water), in feet per day;

- n_e is the effective porosity (connected pores capable of transmitting flow); and

- $\frac{dH}{dX}$ is the hydraulic gradient (change in hydraulic head H, in feet, over distance X, in feet).

Using an average horizontal hydraulic gradient of the Vienna area of 160 ft/mi to the east and 50 ft/mi to the west, a horizontal *K* value of 5.0 ft/d, and porosity of 0.10, the average linear horizontal groundwater velocity in shallow groundwater in the Vienna area is about 1.5 ft/d or about 550 feet per year (ft/yr) to the east and about 0.50 ft/d or about 180 ft/yr to the west. This estimate is for the entire aquifer thickness and assumes a homogenous isotropic aquifer, a condition that is not met in the bedrock aquifer. Using a vertical gradient of 0.02 ft/ft measured in the deep monitoring well MW-08 open to the same interval as public-supply well COV-03, a maximum vertical *K* value of 0.10 ft/d used in models of the Ozark aquifer (Richards, 2010; Imes, 1989), and porosity of 0.10 for the entire thickness of the Ozark aquifer, the fastest estimated linear vertical groundwater velocity in the vicinity of the Vienna Wells site is about 0.02 ft/d or about 7.4 ft/yr.

3.4.2 Distribution of Tetrachloroethene and Other Contaminants of Concern in Groundwater

Analyses of groundwater samples collected from wells at the Vienna Wells site indicate the presence of a plume of PCE in groundwater below the former hat factory property that extends west and north to public-supply wells COV-01 and COV-03, south to well MK-01, and east to the East Tributary (figs. 3-07 and 3-08). Of all the domestic wells sampled, PCE was only detected in one well (MK-01; appendix D, fig. D-12). Within the plume, PCE concentrations range from less than 11 µg/L in wells COV-01, MW-01, MW-02D, MW-08A, MW-09S, MW-09D, and JW-SEEP to more than 500 µg/L in wells MW-03D, MW-06, and MW-06A. The largest PCE concentrations were in two on-site clustered monitoring wells (well clusters MW-06 and MW-03) on and in the immediate vicinity of the former hat factory (fig. 3-07). Moving outward from the former hat factory, PCE concentrations decrease substantially in wells open to shallow groundwater (figs. 3-05 and 3-06), dropping to less than about 50 µg/L about 120 ft to the north (wells MW-04S and MW-04D) and about 230 ft to the south (wells MW-01 and MW-01A) and dropping to less than 5 µg/L about 350 ft to the northeast (well MW-02D; fig. 3-08). In deeper wells, PCE concentrations seem to have spread further, with PCE concentrations between 5 and 50 µg/L in well MK-01 about 600 ft south of the former hat factory and PCE concentrations between 50 and 100 µg/L about 900 ft to the east in wells MW-05S and MW-05D. The large concentrations (greater than 50 µg/L) of PCE in groundwater at monitoring well cluster MW-05 and the general groundwater flow direction to the

northeast suggest that the plume is elongated to the East Tributary, although more wells would be needed to definitively conclude whether the plume is elongated to the east or if two separate plumes exist; however, PCE concentrations in soil samples suggest that there are no sources of PCE in the vicinity of monitoring well cluster MW-05.

Figure 3-07. Maximum concentrations of tetrachloroethene (PCE) in groundwater and surface-water samples collected during 2012-2016, Vienna, Missouri.

The few wells open to deeper groundwater (wells COV-01, COV-03, and MW-08) indicate very little PCE contamination at depth, and it is presumed that, given the transducer data in monitoring well MW-06 (appendix D, figs. D-10 and D-11) and the cone of depression induced by public-supply well COV-03 (figs. 2-05 and 2-06), the extent of the groundwater plume at depth (less than about 710 ft elevation) to the north and west of the former hat factory property is captured by the pumping of public-supply wells COV-01 (before 2012) and COV-03 (from 1994 to the present [2017]). The extent of the plume at depth to the south and southeast is unknown but will likely be captured by public-supply wells COV-01 and COV-03. More deep wells would need to be drilled between the monitoring wells and public-supply wells in the vicinity of the former hat factory property to verify this spatial distribution of PCE in deeper groundwater. The extent of the plume in the shallow subsurface (less than about 710 ft elevation) is more affected by topography and may not be captured by the pumping of public-supply wells COV-01 and COV-03.

Results of geophysical logging and discrete sampling of well MK-01 indicates some PCE contamination is being transported into well MK-01 from around the casing at around 80 ft bgs. Under ambient conditions, downward flow was observed in domestic well MK-01 from below the bottom of casing, and PCE was detected just below the casing indicating a shallow source of PCE (appendix G, fig. G-01). Downward flow increased with depth from about 0.14 gpm at 84.5 ft bgs to about 0.25 gpm at 113.3 ft bgs. Downward flow then slightly decreased to around 0.23 gpm to the bottom of the well. The specific conductance log also showed water with larger salinity entering the well around the casing and decreasing, especially under pumping conditions, as water moved downward. Under pumping conditions, PCE concentrations were largest just below the bottom of casing

(11.08 µg/L; appendix B, table B-01) and decreased with depth to 5.68 µg/L. Concentrations of TCE essentially remained the same in all samples and ranged from an estimated 0.28 to 0.36 µg/L.

Results of geophysical logging and discrete sampling during ambient flow and pumping of well MW-08 indicated no detectable PCE concentrations through the open interval from 90 to 325 ft bgs (appendix G, figs. G-02 and G-03). The caliper log indicated the bottom of the well casing was at about 91 ft bgs and the borehole was rough and had numerous void spaces and vugs ranging from 1 to about 2.7 in. The heat-pulse flowmeter log completed under ambient conditions did not indicate detectable flow. The heat-pulse flowmeter log under pumping conditions, completed on September 22, 2015 (fig. G-03), indicated substantial upward flow only about 13 ft below the pump, which was set at 200 ft bgs. Temperature and specific conductance logs also did not suggest any substantial flow in the well under ambient conditions and very few inflections indicated flow under pumping conditions.

Although PCE was the most frequently detected VOC in groundwater samples and detected in the greatest concentrations (appendix B, tables B-01 and B-07); TCE was also often detected in groundwater samples at small concentrations (appendix D, fig. D-04). The largest TCE concentrations were detected in samples with the largest PCE concentrations; however, concentrations of TCE were low overall and never exceeded the MCL. The degradation product *cis*-DCE was also detected in a subset of samples (tables B-01 and B-07). Concentrations of *cis*-DCE were largest in wells MW-03D, MW-06A, and MW-06, but were also detected at lower concentrations in wells JW-01, COV-01, COV-03, MK-01, MW-02D, and MW-03S. These detections of TCE and *cis*-DCE in samples with largest PCE concentration suggest that the original PCE solvent used at the site was contaminated with TCE and *cis*-DCE or that some reductive dechlorination of PCE is occurring. Concentrations of CFC-113 were largest in MW-03D but were also between 10.1 and 100 µg/L at locations containing large concentrations of PCE in the vicinity of the former hat factory property (monitoring wells MW-01, MW-04D, MW-05S, MW-06, MW-07; cistern JW-01; and spring WP-020; fig. D-05).

Other contaminants of concern identified in groundwater during the BRA included benzene, chromium, and dieldrin (appendix D, figs. D-06 through D-08). Concentrations of benzene, although generally small, were largest in well MW-03S (4.19 µg/L; appendix B, table B-07) that were below the MCL but above a total cancer

risk of 1×10^{-6} level of 0.45 µg/L (fig. D-06). Benzene was also detected at smaller concentrations in wells MW-01 (estimated 0.145 µg/L), MW-02D (estimated 0.013 µg/L), and MW-04D (0.044 µg/L) that were below the total cancer risk level of 1×10^{-6}. The presence of benzene cannot be attributed to any known historical use of chemicals at the former hat factory and is likely from another source, such as spillage from a delivery truck because of the proximity of monitoring well cluster MW-03 to the loading docks. Chromium was detected in samples collected from sites TB2-1 (estimated 2.6 µg/L), TB2-2 (estimated 1.5 µg/L), and WP-020 (estimated 1.4 µg/L) above the total cancer risk of 1×10^{-6} of 0.035 µg/L (fig. D-07). Dieldrin was detected in only one sample from well MW-03D (estimated 0.004 µg/L) above the total cancer risk of 1×10^{-6} of 0.0017 µg/L (fig. D-08).

3.4.3 Trends in Tetrachloroethene and Other Volatile Organic Compounds in Groundwater

Visual inspection of groundwater data and the Mann-Kendall trend test indicated no trend with PCE concentrations over time at any of the groundwater monitoring wells, springs, or seeps from 2012 to 2016, likely because there are so few samples with PCE detections (appendix D, figs. D-02 and D-03); however, PCE concentrations in public-supply well COV-03 before treatment significantly (n=74; probability value [p-value]=0.00000) trend upward from 1997 to 2016 (fig. 1-03). One sample without a detection of PCE was given the value of the reporting limit and included in analysis. The Mann-Kendall trend test requires at least four data points, and many of the groundwater locations had less than four (well JW-01-Cistern, MW-08, MW-08A, MW-09S, MW-09D, MW-05D, MW-05S, MW-06, MW-06A, and MW-07). Likewise, there was not a significant trend with time in TCE concentrations at any of the groundwater sites. Once more groundwater samples are collected, testing for significant trends can be done.

3.4.4 Estimated Mass of Tetrachloroethene in Groundwater

The geographical information system program ArcMap™ (Esri, Redlands, California) was used to make interpolated surfaces of estimated PCE concentrations in groundwater and to estimate the volume of contaminated groundwater and total mass of PCE in shallow (above 710 ft in elevation) groundwater (fig. 3-08). A 4.7-m grid of estimated PCE concentrations in the Ozark aquifer was created from groundwater data

(monitoring wells, domestic wells, and springs) using a nearest-neighbors routine. Additional data points were added to the plume area to set a boundary of 0 µg/L in the aquifer and in the East Tributary (fig. 3-08). A second 2.2-m grid of contaminated aquifer thickness was created for the local shallow flow system, and the two grids were multiplied to calculate PCE mass within each concentration interval and total PCE mass. A porosity of 10 percent was used in the calculations.

The thickness of the shallow aquifer was calculated using measured water levels and the former hat factory property potentiometric map (appendix D, fig. D-01) down to about 710 ft referenced to the NAVD 88. Based on the above assumptions regarding thickness of contaminated zones and a porosity of 10 percent, the total mass of PCE in shallow groundwater plume above the MCL was estimated at about 4.5 (kg, or 9.9 lb covering an area of about 12 acres. Large uncertainty in PCE concentrations and mass exist to the west where no data are available in shallow groundwater (fig. 3-08). The mass of PCE in deeper groundwater could not be estimated because of a lack of data; however, PCE concentrations and monthly pumping data obtained from public-supply wells COV-01 and COV-03 were used to estimate the total mass of PCE removed from groundwater in the 20 years between July 1995 (first groundwater sample with PCE below the MCL in public-supply well COV-01) to June 2015 to be about 7.1 kg or 16 pounds (table D-02), and the total mass of PCE in deeper groundwater will be greater because PCE concentrations are increasing (fig. 1-03).

Figure 3-08. Estimated spatial distribution of tetrachloroethene (PCE) concentrations in shallow groundwater above an altitude of 710 feet at the Vienna Wells site, Vienna, Missouri.

3.5 Distribution of Tetrachloroethene and Other Volatile Organic Compounds in Septic Systems and Sanitary Sewers in the Vienna Area

Wastewater samples collected from sewers during January 2010, February 2012, and March 2012 had small PCE detections from three locations (sites MH-037W, MH-090S, MH-090E, MH-090N, and MH-091N; fig. 3-09; appendix B, table B-01) and indicate that PCE is in the area between sites MH-090 and MH-091 and

seems to be absent to the east in the sewer system upstream from the area. Site MH-037W, located on the east side of Vienna near a marshy area where small springs were identified during the January 2010 seepage run, had a PCE concentration of 0.5 µg/L in a sample analyzed by the portable GC but did not have any PCE detections in subsequent samples collected during February and March 2012. Site MH-090, near the intersection of Highways 63 and 42 and adjacent to a gas station, had three incoming sewer lines intersecting at the junction box (sites MH-090N, MH-090E, and MH-090S). Samples from these three locations and site MH-091N upstream from site MH-090 generally had PCE concentrations around 1 µg/L. Laboratory samples from sites MH-069S, MH-090N, and MH-090E in March 2012 had estimated PCE concentrations of 0.20, 1.0, 0.91 µg/L, respectively (table B-07), and PCE was not detected in the effluent sampled from site MH-064. The PCE detections from sites MH-090S, MH-090E, MH-090N, and MH-091N indicate a PCE source north of MH-090. The absence of detections downstream from the former hat factory property indicates that contaminants are not likely migrating from the former hat factory property into the sewer system.

Of the eight wastewater samples collected from septic and sewer lines (seven portable-GC samples and one laboratory sample) on the former hat factory property, one sample, collected at site Septic South-2 from soil deposited inside the 6-in. sewer pipe on the south side of the former hat factory leading to the city sewer manhole site MH-067, had a PCE concentration of 19.2 µg/kg according to the portable GC (appendix B, table B-02; figs. 2-04 and 3-03). All other samples collected from the sewer lines did not have PCE detections, and the two samples collected from the septic tank north of the former hat factory and from the drainage field of the septic tank did not have PCE detections. Although PCE was detected in several soil samples adjacent to the collapsed sewer line near the central dump, only a small amount of PCE was detected near the surface (table B-02; appendix I, figs. I-10 through I-12) and the lines were less than 1 ft bgs; therefore, the PCE detections near the collapsed sewer line may be from the migration of PCE from the south part of the former hat factory through the septic system but also could be from the central dump or, less likely, the result of contaminated groundwater from the source below the building slab potentially transported through fractures.

Figure 3-09. Concentrations of tetrachloroethene (PCE) in portable gas chromatograph (GC) and laboratory

volatile organic compound (VOC) sewer samples, Vienna, Missouri, 2012.

3.6 Distribution of Tetrachloroethene and Other Volatile Organic Compounds in the Vienna Water-Supply Distribution System

Tetrachloroethene was present in all distribution-system samples collected during the SI ranging from 1.40 to 3.85 µg/L. None of the three water samples collected during subsequent (2014) sampling of water from the distribution system (fig. 1-02) contained detectable concentrations of PCE. The absence of PCE in the 2014 samples demonstrates the efficacy of the treatment system installed on well COV-03 in 2012 after SI activities. Several disinfection by-products were detected in the distribution samples collected during RI activities including bromodichloromethane, dibromochloromethane, dichloromethane, and trichloromethane. All detections were below or slightly above the reporting limit. Concentrations in all samples were below the USEPA MCL for trihalomethanes of 80 µg/L and were below the MCL goal of 7 µg/L (appendix B, table B-07).

3.7 Distribution of Tetrachloroethene and Other Contaminants of Concern in Vapor-Intrusion Samples in the Vienna Wells Area

Tetrachloroethene was detected in several indoor-air and subslab or crawlspace samples at concentrations ranging from 0.46 to 12.0 µg/m^3. Small concentrations (below the 1×10^{-6} cancer risk of 11.0 µg/m^3) of PCE in vapor-intrusion samples suggest little off-site migration of PCE in the shallow subsurface to the east, west, and south (appendix J, fig. J-01). Because of the sensitivity of the analytical method and the widespread use of PCE and other VOCs in a variety of consumer products and industrial materials, indoor-air samples sometime contain larger concentrations of PCE and other VOCs than subslab or crawlspace samples, indicating possible contamination from indoors (USEPA, 2015). For example, the concentration of 1,2-dichloroethane in an indoor-air sample at site VIN-VI-5 on January 13, 2015 was 10.0 µg/m^3; however, the concentration of 1,2-dichloroethane in a crawl-space sample collected on the same day was below the reporting limit of 0.85 µg/m^3 (appendix J, table J-01). At site VIN-VI-10, PCE was detected in the indoor-air sample (0.475 µg/m^3) but was not detected in the crawlspace sample on multiple occasions (fig. J-01).

Only one indoor-air sample collected from site VIN-VI-16 contained PCE (12.0 µg/m^3) larger than a total cancer risk level of 1×10^{-6} (11.0 µg/m^3). This site is north of the former hat factory property and also had PCE detections in the crawlspace.

Other contaminants of concern with exceedance of the total cancer risk level of 1×10^{-6} include benzene and TCE (appendix J, figs. J-02 and J-03). Out of the 17 homes, all had at least 1 sample exceeding the 1×10^{-6} cancer risk threshold for benzene (0.36 µg/m^3), and 5 homes had samples exceeding the 1×10^{-5} cancer risk threshold (3.6 µg/m^3; fig. J-03). Although all five homes with exceedances of the 1×10^{-5} cancer risk seem to have indoor sources of benzene, the widespread occurrence of benzene above the 1×10^{-6} cancer risk in indoor-air samples and in water samples from the Vienna Wells site suggest the former hat factory property is also a source of benzene (USEPA, 2017). Additionally, samples from two homes had TCE concentrations exceeding the 1×10^{-6} cancer risk threshold (4.8 µg/m^3; fig. J-02). One location (site VIN-VI-2) had indoor-air and subslab samples exceeding the threshold, and the other location only had a subslab exceedance. Large concentrations of TCE in two homes adjacent to the former hat factory property suggest a separate TCE source or potential degradation of PCE into TCE. Based on the vapor-intrusion data and multiple lines of evidence described in the vapor-intrusion summary report, the USEPA considered the vapor-intrusion assessment complete and recommended no further action (USEPA, 2017).

3.8 Summary of Contaminant Sources Investigated

Two springs (springs WP-020 and JW-Seep) were identified as containing PCE concentrations, indicating the migration of PCE in shallow groundwater east of the former hat factory property (appendix D, fig. D-09). Shortly thereafter, a site-wide tree-core reconnaissance of 488 trees resulted in the detection of three clusters of trees containing large concentrations of PCE (appendix C, fig. C-01). One cluster south of the former hat factory contained the largest concentrations of PCE on the property. The second cluster, northeast of the former hat factory, is near the metal dump and the former location of a shed that, according to former employees, was used to store chemicals (appendix A, figs. A-01 and A-02). The third cluster was at the eastern property boundary about 200 ft west of spring WP-020 and near the eastern dump.

No pattern was observed indicating residual PCE in the Vienna sanitary sewer system derived from the former hat factory property. Septic and sewer-line samples collected on the former hat factory property (fig. 3-03) contained less than 20 µg/kg of PCE, providing minimal indication of contamination derived from these source areas.

A series of groundwater monitoring wells were installed at the Vienna Wells site at various depths to ascertain groundwater quality and velocity (fig. 3-07). The largest groundwater concentrations of PCE were underneath and in the immediate vicinity of the former hat factory, and concentrations decreased with increasing distance away from the former hat factory with the exception of clustered well MW-05S on the east part of the former hat factory property near the eastern dump. Although a separate source of PCE at the eastern dump is possible, it is not probable because PCE concentrations in soil samples in the vicinity of well cluster MW-05 were small or below detection. Local groundwater at the former hat factory property was determined to flow primarily to the northeast on the west side of the property, likely because of the influence of pumping from public-supply well COV-03, and transitions more eastward in areas east of the property where it emerges as a spring at site WP-020 near the East Tributary.

Results of the soil sampling indicated the cluster of large PCE concentrations below and in the immediate vicinity of the former hat factory and near the central dump; however, substantial concentrations of PCE were not detected in soil samples collected in the vicinity of the eastern dump. Initially these three clusters of PCE detections seemed to be separate sources of PCE; however, groundwater direction and the dramatic change in elevation between the western side (843 ft) and the eastern side (781 ft) of the property where groundwater emerges at a spring indicate that PCE may be easily transported to tree roots near the source (former hat factory) and where groundwater nears the surface (east of the former hat factory property).

4.0 Contaminant Fate and Transport at the Vienna Wells Site

4.1 Soils

Based on the preponderance of evidence presented in this report, the primary source of PCE at the Vienna

Wells site is underneath and in the immediate vicinity of the former hat factory building. The amount of contamination is estimated to be 33 kg (about 73 lb) of PCE present in soils and 4.5 kg (about 9.9 lb) present in groundwater. Although PCE-contaminated waste might have been dumped into the floor drains, the general absence of PCE in soils along the ditch west of the former hat factory and in the sanitary sewer system suggests that this was not a major route of transport of PCE contamination (fig. 3-03). The mechanism of contamination of these soils, therefore, is interpreted to be possible seepage through cracks in the slotted floor and dumping south of the building.

An examination of the vertical profiles of PCE in soils indicates that PCE concentrations in the upper 2 ft of soil are typically substantially smaller than those at greater depths (appendix I, figs. I-10 through I-19), with the exception of samples collected on the south side of the former hat factory. In most areas, volatilization from the shallow subsurface probably is an important loss mechanism within the upper few feet of soil at the site and volatilization loss of PCE and other VOCs from shallow (less than about 2 ft deep) soil probably contributes to the increase in PCE with depth in many samples less than a few feet bgs.

The primary loss or degradation mechanism for PCE in the subsurface is biodegradation by reductive dechlorination. Microbial reactions occur throughout the subsurface if there is sufficient moisture, an organic carbon source, and appropriate electron acceptors. Historically, PCE was used to remove animal fats from animal furs. Animal fats are sources of organic carbon and, under favorable conditions, microorganisms can readily couple the oxidization of these fats and existing organic carbon in soils with the reduction of PCE and TCE. The reductive dechlorination of PCE requires anaerobic conditions (preferably iron or sulfate reducing), which can exist as "microenvironments" within the pores in soils. The large moisture content present in the highly clayey soils in the area also substantially contributes to the development of anaerobic conditions.

A plot of the molar fraction of PCE to TCE and *cis*-DCE in soil samples with detections indicates that some microbial degradation of PCE by reductive dechlorination is occurring (fig. 4-01), although most samples are greater than 80 percent PCE. Six soil samples plot near the center or near the base of the diagram, indicating moderate amounts of biodegradation of PCE. These samples contained a molar fraction of PCE between about 11 and 63 percent. The fractions of TCE or *cis*-DCE in these samples were substantially larger than can be attributed

to these compounds present as 1 or 2 percent impurities in commercial-grade PCE (USEPA, 2003). Overall, the distribution of PCE, TCE, and *cis*-DCE in soil samples indicates that moderate reductive dechlorination has occurred in about one-third of the samples. On average, more dechlorination seems to be occurring in the unconfined, grassy area A (sites A017, A018, A024, A028, A029, and A052) where the average PCE composition is 69 percent compared to samples from confined area under the slab (sites D024, D028, D029, D030, D033, D035, D048, D050, D051, D058, D059, D061, and D066) where the average PCE composition is about 88 percent. This difference is likely due to increased soil moisture in area A and limited soil moisture under the concrete slab. Abiotic reductive dechlorination, such as hydrolysis, is slower than biotic reductive dechlorination, but likely more dominant in more aerobic environments, such as under the former hat factory slab and at shallow depths below ground.

Another major loss mechanism of PCE is transport from overlying soils to groundwater. With an estimated 33 kg of PCE in soils, 4.5 kg of PCE in shallow groundwater, and 7.1 kg of PCE removed by public-supply well COV-03, about 11.6 kg in total, or more than 25 percent of the total PCE deposited at the Vienna Wells site, has migrated into groundwater. Because there is a large amount of uncertainty in the original amount of PCE deposited at the site and the estimation of PCE in groundwater, this percentage could be substantially larger or smaller.

Although volatilization and some reductive dechlorination of PCE in contaminated soils have occurred, these processes seem to be slow and incomplete as complete reductive dechlorination produces ethane, which is not present, and after potentially 48 to 65 years residence time in soils based on a probable release between 1952 and 1996 (table 1-01), PCE still composes almost 90 percent of the total chlorinated VOCs in the vicinity of the former hat factory. Because substantial PCE concentrations remain in soils under the concrete slab of the former hat factory, mechanisms for PCE loss from soil will not remove a substantial mass of the remaining PCE from the soils for decades to come, and these soils will be a continued source of PCE to groundwater.

Figure 4-01. Percentage of tetrachloroethene (PCE), trichloroethene (TCE), and *cis*-1,2-dichloroethene (*cis*-DCE) by molar weight in soil samples, Vienna, Missouri, 2012–14.

4.2 Groundwater

Groundwater beneath the Vienna Wells site generally is aerobic with dissolved oxygen concentrations averaging 4.9 milligrams per liter (mg/L) and ranging from about 0.2 to 8.9 mg/L (appendix B, table B-07). Samples from wells COV-01, COV-03, MK-01, and MW-05D had among the smallest dissolved oxygen concentrations with an average of less than 0.4 mg/L. Overall, dissolved oxygen concentrations tended to decrease in wells with increasing completion depth. According to the USEPA Natural Attenuation Screening Protocol, concentrations of dissolved oxygen larger than 0.5 mg/L in groundwater generally indicate aerobic conditions that are not favorable for the reductive dechlorination of PCE or TCE (USEPA, 1998a). Aerobic conditions, however, are conducive to the degradation of *cis*-DCE through oxidative pathways. The general absence of large concentrations of TCE and *cis*-DCE as compared to PCE concentrations in groundwater samples indicates that substantial biodegradation of PCE by reductive dechlorination is not occurring (table B-07). The absence of substantial concentrations of daughter products is consistent with the detection of dissolved oxygen concentrations in the milligrams per liter range in most groundwater samples.

A plot of the molar fraction of PCE to TCE and *cis*-DCE in groundwater samples that had detectable concentrations of one or more of these compounds by the laboratory indicates that overall, minimal biodegradation of PCE is occurring in groundwater (fig. 4-02). For all molar fraction calculations in groundwater, censored values were set to equal the value of the reporting limit. All but 5 of the 99 samples plot near the upper vertex of the diagram, indicating that more than 70 percent of the total chlorinated ethenes (PCE plus TCE plus *cis*-DCE) are present as the parent compound PCE. Microbial degradation of PCE is not substantial in groundwater at this site.

Figure 4-02. Percentage of tetrachloroethene (PCE), trichloroethene (TCE), and *cis*-1,2-dichloroethene (*cis*-DCE) by molar weight in groundwater samples, Vienna, Missouri, 2012-2014.

In the absence of substantial biodegradation, the factors most important in the fate and transport of PCE in groundwater are advection, hydrodynamic dispersion, retardation by sorption, and hydrolysis. Advection is the

bulk movement of groundwater and its dissolved contaminants, and is the primary process that moves dissolved contaminants through groundwater. Using Darcy's Law, the average linear vertical groundwater velocity in groundwater at the former hat factory property was estimated to be about 7.4 ft/yr. As was discussed, steep hydraulic gradients in the vicinity of the former hat factory indicate that the horizontal K value may be much larger than the vertical K value; however, there are areas, such as well cluster MW-03, where vertical gradients are smaller and vertical transport of groundwater may be greater. Because public-supply wells COV-01 and COV-03 are open at about 385 and 321 ft bgs, respectively, and migration from the former hat factory property to the public-supply wells is primarily limited by vertical transport because of anisotropy, it would take about 53 and 44 years for groundwater to migrate from the source area underneath the former hat factory to public-supply wells COV-01 and COV-03, respectively. The earliest probable PCE disposal at the former hat factory is 1965 (table 1-01), about 29 years before the first measured PCE concentration of 0.90 µg/L in public-supply well COV-01 and 32 years before the first measured PCE concentration of 0.50 µg/L in public-supply well COV-03 (fig. 1-03). Because pumping of the public-supply wells may substantially decrease the total transport time, the general agreement in the estimated transport time from the former hat factory property and the initial occurrence of PCE in the two public-supply wells supports the conclusion that contamination from the former hat factory property is the cause of increased PCE concentrations in both public-supply wells. The large mass of PCE in soil (33 kg) compared to groundwater (7.1 kg pumped from 1995 to 2015 and 4.5 kg currently [2017] in groundwater) coupled with a lack of degradation indicate that contamination will likely persistent over many decades.

Hydrodynamic dispersion will decrease PCE concentrations through mechanical and chemical (diffusion) processes. Because the organic content of the underlying bedrock aquifer is likely low (less than 0.1 percent; Adamski, 1997), retardation of PCE migration by sorption or partitioning onto aquifer solids probably is small; however, some retardation may occur from migration of PCE into fractures and vugs. Rates of hydrolysis, another potential factor affecting PCE fate, vary substantially from about 1 to 2 up to more than 10 years (USEPA, 2008). Although hydrolysis generally has a minor effect on chemical fate relative to microbial degradation, when microbial degradation is unsubstantial, hydrolysis can be an important factor in chemical fate.

4.3 Surface Water

Volatilization is the primary mechanism affecting the fate and transport of PCE detected in the east and west tributaries at the Vienna Wells site. Because only one sample from a small trickling stream contained an estimated PCE concentration of 2.2 µg/L (site TB2-Trickle), PCE, if present during rainfall, will quickly volatilize.

5.0 Baseline and Ecological Risk Assessment For the Vienna Wells Site

A Baseline Human-Health Risk Assessment and an Ecological Risk Assessment were completed for COPCs and preliminary COPC (appendix L, tables L-01 through L-04).

5.1 Baseline Human-Health Risk Assessment

The HHRA was completed by Black and Veatch Special Projects Corporation (Black and Veatch Special Project Corporation, 2016). The purpose of the HHRA was to evaluate the potential threat to human health from the release or potential release of COPCs at the Vienna Wells site and to evaluate the need for remedial action based on whether or not the site presents risks greater than the acceptable range. Methods used in the HHRA are outlined in USEPA (1989). Media evaluated for the HHRA included surface soil, subsurface soil, sediment, surface water, groundwater, soil-gas, and indoor-air samples collected during field investigations from 2007 through 2013. Only laboratory analytical data were used in the HHRA.

The major source of PCE and other COPCs identified for the HHRA was PCE in soils and COPCs migrating to groundwater from those soils. The complete exposure pathways identified for evaluation in the HHRA were:

- Current/future quantitative evaluation of exposure to soil at on-site source areas at the Vienna Wells site through incidental ingestion and dermal absorption (lifetime and child residents, adult industrial/commercial workers, adolescent recreational users, and adult construction workers);

- Current/future quantitative evaluation of exposure to dust and vapors at on-site source areas at the Vienna Wells site through inhalation of dust and vapors (lifetime and child residents, adult industrial/commercial workers, adolescent recreational users, and adult construction workers);

- Current/future quantitative evaluation of exposure to groundwater at on-site source areas at the Vienna Wells site through direct ingestion (lifetime and child residents and industrial/commercial workers);

- Current/future quantitative evaluation of exposure to groundwater at on-site source areas at the Vienna Wells site through dermal absorption (lifetime and child residents)—Exhibit 1-2 in the Risk Assessment Guidance for Superfund Part E (USEPA, 2004) will be used to determine whether or not each COPC should be quantitatively assessed for the dermal exposure route;

- Current/future quantitative evaluation of exposure to groundwater at on-site source areas at the Vienna Wells site through inhalation of vapors associated with the use of groundwater (lifetime and child residents);

- Current/future quantitative evaluation of exposure to groundwater at on-site source areas at the Vienna Wells site through inhalation of vapors associated with indoor vapor intrusion (lifetime and child residents, recreational users/trespassers, and industrial/commercial workers);

- Current/future quantitative evaluation of exposure to groundwater at on-site source areas at the Vienna Wells site through incidental ingestion and dermal absorption, if the depth to groundwater is such that it could pool in an excavated trench (construction workers);

- Current/future quantitative evaluation of exposure to groundwater at on-site source areas at the Vienna Wells site through inhalation of vapors in an excavated trench (construction workers; the Virginia Department of Environmental Quality trench model will be used to evaluate this pathway [Black and Veatch Special Projects Corp., 2016]);

- Current/future quantitative evaluation of exposure to groundwater at offsite residential properties through direct ingestion and dermal absorption (lifetime and child residents);

- Current/future quantitative evaluation of exposure to groundwater at offsite residential properties through inhalation of vapors associated with the use of groundwater and indoor vapor intrusion (lifetime and child

residents);

- Current/future quantitative evaluation of exposure to surface water downstream from the Vienna Wells site through ingestion and dermal absorption (adolescent recreational users); and

- Current/future quantitative evaluation of exposure to sediment downstream from the Vienna Wells site through ingestion and dermal absorption (adolescent recreational users).

The characterization of risk from the site integrates the results of an exposure and toxicity assessment to derive quantitative and qualitative estimates of the potential for adverse health effects that may occur because of exposure from site-related COPCs. The potential adverse health effects were classified as noncarcinogenic (noncancer) and carcinogenic (cancer) effects (response). The potential for a chronic or subchronic noncarcinogenic health effect is expressed as a hazard quotient (HQ). The HQ is a unitless ratio of the chronic or subchronic daily intake or dermal absorbed intake divided by a reference dose that is a threshold below which no adverse health effects will be seen (USEPA, 1989). If the HQ exceeds 1.0 (indicating the site exposure exceeds the threshold dose), then concern may exist for potential noncancer health effects. Although a HQ above 1.0 indicates a greater level of concern, the HQ is not a statistical probability. Unlike the noncancer hazard (HQ), the potential for adverse cancer risk is expressed as a probability; for example, a cancer risk of 1×10^{-4} is equivalent to 1 cancer case in 10,000 exposed people, and a cancer risk of 1×10^{-6} is 1 case in 1 million exposed people. The USEPA considers calculated risks between 1×10^{-6} and 1×10^{-4} to be in the acceptable range.

Exposures to future on-site residents were evaluated for surface soil (ingestion, dermal contact, and inhalation of dust), combined surface and subsurface soil (inhalation of outdoor vapors), groundwater (ingestion, dermal contact, and inhalation of vapors from household use of water), and for potential indoor vapor intrusion pathways based on detected COPC. Risks were characterized separately for each of 15 hypothetical residential exposure areas and for potential future site-wide exposures (appendix L, fig. L-01; table L-04). Additionally, exposures to current and future off-site residents were evaluated for groundwater (ingestion, dermal contact, and inhalation of vapors from household use of water), and potential indoor-vapor intrusion pathways. Risks were also characterized separately for each off-site residential well that was sampled during the RI (table L-04).

Based on the completed exposure pathways and calculations made in the HHRA, the Vienna Wells site

presents an unacceptable cancer risk and noncancer HQ to future residents and workers. Sitewide, the estimated cancer risk to future residents is 5×10^{-4} because of potential ingestion and dermal contact with soil containing benzo[*a*]anthracene, benzo[*a*]pyrene, benzo[*b*]fluoranthene, dibenzo[*ah*]anthracene, arsenic, and chromium; groundwater with benzene, dieldrin, PCE, TCE, and chromium; and vapor intrusion containing benzene, PCE, and TCE. The site-wide HQ value was 32 because of ingestion and dermal contact with soil containing manganese, groundwater containing PCE, and vapor intrusion containing PCE. Also an exposure affecting the HQ value was potential ingestion of soil containing arsenic and thallium. Because of potential ingestion of, dermal contact with, and inhalation of groundwater containing PCE and vapor intrusion containing PCE, the off-site residential well JW-01 had an HQ value of 4. Several off-site residential homes had HQ values exceeding 1 (ranged from 0.2 to 7) because of vapor intrusion of various VOCs (appendix L, table L-04). One location (well VIN-VI-8; shown as Off Site Residential at 545 North Chestnut in table L-04) had an excessive risk of cancer (8×10^{-4}) because of concentrations of 1,2-dichloroethane, benzene, bromodichloromethane, carbon tetrachloride, and chloroform, although the occurrence of these contaminants at large concentrations are likely the result of contamination from indoor sources. Concentrations of PCE and other COCs detected in surface-water and sediment samples did not present unacceptable risks or hazards to human health. Concentrations of PCE in groundwater represent an unacceptable hazard to future industrial/commercial workers through direct ingestion and inhalation of vapors through the hypothetical indoor vapor-intrusion pathway.

Current (2017) PCE concentrations in the distribution system with the implementation of the air stripper are below the MCL for PCE (fig. 1-03). Based on historic PCE concentrations before treatment, if use of the air stripper were to be discontinued then PCE concentrations in finished water entering the distribution system would likely increase to PCE concentrations before treatment, which are above the MCL.

5.2 Ecological Risk Assessment

An ERA was completed as part of the RI/FS effort to evaluate the ecological risks at the Vienna Wells site. A complete assessment of the ecological risks for the Vienna Wells site is in the Baseline Risk Assessment Vienna Wells Site, Maries County, Missouri (Black and Veatch Special Projects Corp., 2016). The methodology

used in the ERA was based on, and complied with, USEPA (1997) and (1998b).

Data collected from 2007 through 2015 was used for screening and included 102 soil samples (2 surficial soil samples from MDNR collected during the SI, 14 surficial soil samples collected by the USGS during the RI, and 86 surficial soil samples collected by the USEPA in March 2015), 6 surface-water samples collected the USGS as part of the RI, 6 sediment samples collected by the USGS as part of the RI, and numerous groundwater samples from 12 monitoring wells.

Ecological screening values are contaminant-specific, media-specific concentrations that are used to determine if any particular contaminant might pose a risk to ecological receptors. Concentrations of the full list of constituents on the HSL (appendix K, table K-01) were compared to ecological screening values to determine the risk that contaminants in each media might pose to ecological receptors. Tables of all the preliminary COPCs in the various media are in appendix L, tables L-01 through L-03.

Based on the refinement screening completed by Black and Veatch, *bis*(2-ethylhexyl)phthalate; heptachlor epoxide; methyl acetate; total dichlorodiphenyldichloroethane, dichlorodiphenyldichloroethylene, and dichlorodiphenyltrichloroethane; and lead were all identified as COPCs for terrestrial receptors (soil exposure) (appendix L, table L-01). Likewise for aquatic receptors (water exposure), desethyl atrazine, CFC-113, chloromethane, bromide, and nitrate were COPCs in surface water (refinement table not produced in risk assessment); and *p*-Cresol was a COPC in sediment (appendix L, table L-02). Based on the potential for exposure and adverse effects of each COPC, the potential risk to terrestrial and aquatic receptors was considered to be low and very low, respectively. The evaluation concluded that the overall potential risks to terrestrial and aquatic receptors at the Vienna Wells site are negligible.

6.0 Conclusions for the Vienna Wells Site

Results of the Remedial Investigation of the Vienna Wells site indicate the presence of substantial contamination in soils and groundwater that present unacceptable human-health risks and hazards as determined by the Baseline Risk Assessment completed by Black and Veatch Special Projects Corporation. Based on the preponderance of evidence presented in this report, the primary source area is a tetrachloroethene (PCE) source

area in soils under the slotted-floor section of the former hat factory. Within this area, PCE was detected in only two soil samples above the U.S. Environmental Protection Agency Region 9 residential screening level for soil of 24,000 micrograms per kilogram. Concentrations of PCE in soils generally increased with increasing depth; however, concentrations of PCE decreased with depth in the vicinity of the slotted-floor part of the former hat factory where they were the largest measured at the site. The origin of PCE at the Vienna Wells site was the apparent use of PCE as a solvent in the hat-making process, although there are no records to confirm usage at the site. Because very few samples exceeded the residential screening level for soil, PCE in soil was determined to not present unacceptable cancer risk or noncancer hazard sitewide to future residents and current and future workers; however, concentrations of several polycyclic-aromatic hydrocarbons and metals (arsenic and chromium) pose an excessive cancer risk in soil, and manganese poses an excessive noncancer risk in soil.

Analyses of groundwater samples collected from monitoring wells at the Vienna Well site indicated that a plume of PCE contamination in shallow (greater than about 710 feet [ft] elevation) groundwater extends from the former hat factory to the east to a small creek where it likely discharges in seeps and springs, including site WP-020. Concentrations of PCE in wells MW-01 and MK-01 suggest some migration in groundwater to the south. Within the plume, PCE concentrations in the bedrock ranged from about 1,000 micrograms per liter (μg/L) directly below and in the immediate vicinity of the former hat factory to less than 100 μg/L about 200 ft south of the former hat factory and decreased to less than 10 μg/L about 125 ft north of the former hat factory. The plume extends deeper into the aquifer affecting two nearby public-supply wells that are cased more than 300 ft below ground surface. It is presumed that pumping from these public-supply wells captures most PCE migrating deeper into the aquifer given transducer data from monitoring wells MW-06 and the depression in the water table induced by the pumping of public-supply well COV-03. Although no water-level data are available for public-supply well COV-01, it is likely that pumping caused a comparable depression in the water table before it was put on standby in 2007. Because there is no gap between the pumping of public-supply well COV-01 (from 1954 to 2007) and the beginning of pumping in public-supply well COV-03 in 1994 (fig. 1-03), most contaminated groundwater at depth (less than about 710 ft elevation) at the Vienna Wells site likely has always been captured by one of these two public-supply wells. More deep wells would need to be drilled between the monitoring wells

and public-supply wells in the vicinity of the former hat factory property to verify this spatial distribution of PCE in deeper groundwater. The extent of the plume in the shallow subsurface (greater than about 710 ft elevation) is more affected by topography and may not be captured by the pumping of public-supply wells COV-01 and COV-03. Based on the completed exposure pathways and calculations made in the human-health risk assessment (HHRA), groundwater at the Vienna Wells site presents the largest unacceptable cancer risk of 4×10^{-4} and noncancer hazard quotient of 17 to future residents and workers because of potential ingestion and dermal contact with groundwater with benzene, dieldrin, PCE, trichloroethene (TCE), and chromium.

The area of the groundwater plume containing PCE concentrations above the 5 µg/L maximum contaminant level (MCL) is about 12 acres in shallow groundwater and contains an estimated 4.5 kilograms (kg) or 9.9 pounds of PCE. The mass of PCE in deeper groundwater could not be estimated because of a lack of data; however, PCE concentrations and monthly pumping data obtained from public-supply wells COV-01 and COV-03 were used to estimate the total mass of PCE removed from groundwater in the 20 years between July 1995 (first groundwater sample with PCE below the MCL in public-supply well COV-01) to June 2015 to be about 7.1 kg or 16 pounds. No significant trends with time were observed at any groundwater monitoring wells, springs or seeps from 2012 to 2016 likely because of the short period of record (4 years) for PCE concentrations; however, PCE concentrations in public-supply well COV-03 before treatment significantly trend upward from 1997 to 2016.

Based on the generally aerobic conditions in most of the bedrock aquifer and the predominance of PCE as compared to its degradation products TCE and *cis*-1,2-dichloroethene in groundwater samples, microbial degradation of PCE is not substantial in the bedrock aquifer.

Groundwater flow in the vicinity of the former hat factory is mostly controlled by the pumping of well COV-03, especially in deeper groundwater (less than about 710 ft in elevation), and, to a lesser extent, topography. In contrast, groundwater flow to the east is mainly controlled by topography. Vertical hydraulic gradients estimated from water levels in clustered wells are generally large and indicate that vertical hydraulic conductivity may be much smaller than horizontal hydraulic conductivities; however, some areas, such as near well cluster MW-03, may have small vertical hydraulic gradients. Slug tests completed as part of the Remedial

Investigation indicate horizontal hydraulic conductivities between 0.04 to 50 feet per day in the Roubidoux

Formation and 0.50 to 0.9 feet per day in well MW-08 in the Eminence Dolomite formation. The average

groundwater velocity in the bedrock aquifer at the former hat factory property is about 7.4 feet per year, and it

would take at about 53 and 44 years for groundwater to migrate from the source area underneath the former hat

factory to public-supply wells COV-01 and COV-03, respectively. The earliest probable PCE disposal at the

former hat factory is 1965 (table 1-01), about 29 years before the first measured PCE concentration of 0.90 µg/L

in public-supply well COV-01 and 32 years before the first measured PCE concentration of 0.50 µg/L in public-

supply well COV-03 (fig. 1-03). Because pumping of the public-supply wells may substantially decrease the total

transport time, the general agreement in the estimated transport time from the former hat factory property and the

initial occurrence of PCE in the two public-supply wells supports the conclusion that contamination from the

former hat factory property is the cause of increased PCE concentrations in both public-supply wells. The large

mass of PCE in soil (33 kg) compared to groundwater (7.1 kg pumped from 1995 to 2015 and 4.5 kg currently

[2017] in groundwater) coupled with a lack of degradation indicate that contamination will likely persistent over

many decades.

Current (2017) PCE concentrations in the water distribution system with the implementation of the air

stripper are below the MCL for PCE. Based on historic PCE concentrations before treatment, if use of the air

stripper were to be discontinued then PCE concentrations in finished water entering the distribution system would

likely increase to PCE concentrations before treatment, which are above the MCL (fig. 1-03).

Overall, concentrations of PCE in vapor-intrusion samples were small, and all field blank results were

below reporting limits. Tetrachloroethene was detected in several indoor-air and subslab or crawlspace samples at

concentrations ranging from 0.46 to 12.0 micrograms per cubic meter ($\mu g/m^3$); however, only one indoor-air

sample collected from site VIN-VI-16 contained PCE (12.0 $\mu g/m^3$) larger than a total cancer risk level of 1×10^{-6}

(11.0 $\mu g/m^3$). Small concentrations of PCE in vapor-intrusion samples suggest little off-site migration of PCE to

the north, west, and south, and large concentrations of TCE in two homes adjacent to the former hat factory

property suggest potential degradation of PCE into TCE. Several off-site residential homes had hazard quotients

exceeding 1 and ranged from 0.2 to 7 because of vapor intrusion of various volatile organic compounds. One

location (site VIN-VI-8) has excessive risk of cancer because of concentrations of 1,2-dichloroethane, benzene, bromodichloromethane, carbon tetrachloride, and chloroform.

The ecological risk assessment determined that, based on the potential for exposure and adverse effects of each chemical of potential concern, the potential risk to terrestrial and aquatic receptors was considered to be low and very low, respectively.

7.0 References Cited

Adamski, J.C., Nutrients and pesticides in ground water of the Ozark Pateaus in Arkansas, Kansas,
Missouri, and Oklahoma: U.S. Geological Survey Water-Resources Investigations Report 96-43136,
34 p.

Bachle, Peter, 2009, Geohydrologic summary of Vienna Wells site (former hat factory): Geological
Survey Program, Division of Geology and Land Survey, Missouri Department of Natural Resources,
11 p.

Bienkiewicz, K., 1983, Physical chemistry of leathermaking: Krieger Publishing Co., Malabar, Florida,
541 p.

Black and Veatch Special Projects Corp., 2016, Draft Baseline Risk Assessment, Vienna Wells Site,
Maries County, Missouri: Overland Park, Kansas, 149 p.

Blanc, T.J., Austin, A., Ronald, B., Dierking, J., Kromrey, G., Smith, M., 2001, Missouri Watershed
Inventory and Assessment—Gasconade River: Missouri Department of Conservation, West Central
Region-Fisheries Division: accessed September 2015, at
https://mdc.mo.gov/sites/default/files/watersheds/Gasconade130.pdf.

Czarnecki, J.B., Gillip, J.A., Jones, P.M., and Yeatts, D.S., 2009, Groundwater-flow model of the Ozark
Plateaus aquifer system, northwestern Arkansas, southeastern Kansas, southwestern Missouri, and
northeastern Oklahoma: U.S. Geological Survey Scientific Investigations Report 2009-5148 (Revised
March 2010), 62 p.

Fenneman, N.M., 1938, Physiography of the Eastern United States: New York, McGraw-Hill Book Co.,
p. 419-518.

Imes, J.L., and Emmett, L.F., 1994, Geohydrology of the Ozark Plateaus aquifer system in parts of
Missouri, Arkansas, Oklahoma, and Kansas: U.S. Geological Survey Professional Paper 1414-D, 127

p.

Imes, J.L., 1989, Analysis of the effect of pumping on ground-water flow in the Springfield Plateau and
Ozark aquifers near Springfield, Missouri: U.S. Geological Survey Water-Resources Investigations
Report 89-4079, 63 p.

Jackson, S., 2009, Site Inspection Report, Vienna wells site, Maries County, Missouri: Missouri
Department of Natural Resources, Division of Environmental Quality, Hazardous Waste Program,
MON000705803, 48 p.

Johnson, A. E., 1971, Dry cleaning: Merrow Publishing Company, 48 p.

Limmer, M., Holmes, A., and Burken, J., 2014, Phytoremediation of Chlorinated Ethenes in Trees—A
Four-Year Study of Seasonal Chemodynamics *in Planta*: Environmental Science and Technology, v.
48, no. 18, p. 10634-10640.

Linn, B., 2009, Chemicals used in drycleaning operations: State Coalition for Remediation of
Drycleaners, 17 p.

Meramec Regional Planning Commission, 2014, Maries County Hazard Mitigation Plan—Planning
Area Profile and Capabilities: 28p, accessed January 2015, at
*http://meramecregion.org/Hazard%20Mitigation/Maries%20Co.%20Hazmit%20DRAFT/2Planning
%20area%20profileMaries.pdf*.

Miller, D., and Vandike, J., 1997, Groundwater Resources of Missouri: Missouri Department of Natural
Resources, Rolla, Missouri, 227 p.

Missouri Department of Conservation, 2009, Missouri Department of Conservation Natural Heritage
Review On-Line Level 1 Report: Missouri Department of Conservation Natural Heritage Review
Policy Coordination Unit, Jefferson City, Missouri.

Missouri Department of Health and Senior Services, 2014, Public health assessment—Vienna wells: EPA
MON000705803, accessed September 2015, at

https://www.atsdr.cdc.gov/HAC/pha/ViennaWells/ViennaWellsSite_PHA_Final_10-20-15_508.pdf.

Missouri Department of Natural Resources, 2007, Missouri Environmental Geology Atlas: Rolla,
Division of Geology and Land Survey, CD-ROM.

Missouri Spatial Data Information Service, 2005, Land use land cover: accessed September 2015, at
http://msdis.missouri.edu/data/lulc/lulc05.html.

Morrison, R.D., Murphy, B.L., 2013, Chlorinated solvents—a forensic evaluation: Royal Society of
Chemistry, v. 1, p. 415.

National Oceanic and Atmospheric Administration, 1992, Rainfall frequency atlas of the Midwest:
accessed September 2015, at *http://www.sws.uiuc.edu/pubdoc/B/ISWSB-71.pdf*.

National Oceanic and Atmospheric Administration, 2015, National Climatic Data Center, Climatic Data
Online: accessed January 2015, at *http://www.ncdc.noaa.gov/cdo-web/*.

Pugh, A.L., 2008, Summary of aquifer test data for Arkansas—1940-2006: U.S. Geological Survey
Scientific Investigations Report 2008-5149, 33 p.

Rantz, S.E., and others, 1982, Measurement and computation of streamflow—Volume 1. Measurement
of stage and discharge: U.S. Geological Survey Water-Supply Paper 2175, 284 p.

Richards, Joseph M., 2010, Groundwater-flow model and effects of projected groundwater use in the
Ozark Plateaus Aquifer System in the vicinity of Greene County, Missouri - 1907-2030: U.S.
Geological Survey Scientific Investigations Report 2010-5227, 106 p.

Schubert, A.L., 2001, Missouri Watershed Inventory and Assessment—Osage River east: Missouri
Department of Conservation, West Central Region-Fisheries Division: accessed September 2015, at
https://mdc.mo.gov/sites/default/files/resources/2011/04/300cotxt-1.pdf .

Tetra Tech, Inc., 2015, Field sampling report with data summary regarding remedial soil sampling—
Vienna Wells site, Vienna, Missouri: Superfund Technical Assessment and Response Team, 526 p.

Thompson, T.L., 1991, Paleozoic Succession in Missouri, Part 2, Ordovician System: Rolla, Missouri

Division of Geology and Land Survey Report of Investigation RI-70, 282 p.

United Nations Environment Programme, 1987, Montreal protocol on substances that deplete the ozone layer, accessed June, 2016, at *http://www.unep.ch/ozone/pdf/Montreal-Protocol2000.pdf*.

U.S. Census Bureau, 2010a, American fact finder, accessed January 2015, at *http://factfinder.census.gov/faces/nav/jsf/pages/community_facts.xhtml*.

U.S. Census Bureau, 2010b, 2010 census gazetteer files, accessed January 2015, at *http://www.census.gov/geo/maps-data/data/gazetteer2010.html*.

U.S. Department of Agriculture, 2012, USDA Designates an Additional 39 Counties In 8 States as Primary Natural Disaster Areas Due to Worsening Drought: accessed January 2015, at *https://www.usda.gov/wps/portal/usda/usdahome?contentid=2012/07/0243.xml&contentidonly=true*.

U.S. Environmental Protection Agency, 1989, Risk Assessment Guidance for Superfund: Volume I, Human Health Evaluation Manual, Part A: Office of Solid Waste and Emergency Response, OSWER Directive 9285-7-01.

U.S. Environmental Protection Agency, 1997, Ecological risk assessment guidance for superfund: process for designing and conducting ecological risk assessments-interim final: Doc. No. EPA 540-R-97-006.

U.S. Environmental Protection Agency, 1998a, Technical protocol for evaluating natural attenuation of chlorinated solvents in ground water: U.S. Environmental Protection Agency National Risk Management Laboratory, Office of Research and Development, Cincinnati, Ohio, doc. No. EPA/600/R-98/128.

U.S. Environmental Protection Agency, 1998b, Guidelines for Ecological Risk Assessment—Final: Risk Assessment Forum, Washington, D.C., Doc. No. EPA/630/R-95/002F.

U.S. Environmental Protection Agency, 2003, Focused remedial investigation of operable units OU1 and OU3, Riverfront Superfund Site, Franklin County, Missouri, U.S. Environmental Protection Agency Region 7, contract DW1495217301-2, 128 p.

U.S. Environmental Protection Agency, 2006, Focused remedial Investigation of Operable Unit 5, Old Hat Factory, Riverfront Superfund Site, Franklin County, Missouri: USEPA DW1495217301-2.

U.S. Environmental Protection Agency, 2008, Focused remedial Investigation of Operable Unit 4, Riverfront Superfund Site, Franklin County, Missouri: USEPA DW1495217301-2.

U.S. Environmental Protection Agency, 2010, Hazardous ranking system documentation record—Vienna wells: EPA MON000705803.

U.S. Environmental Protection Agency, 2011a, General scope of work for interagency agreement between the U.S. Geological Survey and the U.S. Environmental Protection Agency—Vienna Wells site, Vienna, Missouri: U.S. Environmental Protection Agency, Region 7, 17 p.

U.S. Environmental Protection Agency, 2011b, Quality assurance project plan for the remedial investigation of the Vienna Wells site, Maries County, Missouri: U.S. Environmental Protection Agency, Region 7, 115 p.

U.S. Environmental Protection Agency, 2014, Phase 3 Sampling Plan for Characterization of Soils, Groundwater, and Surface-water at the Vienna Wells Site, Vienna, Missouri: U.S. Environmental Protection Agency, Region 7, 20 p.

U.S. Environmental Protection Agency, 2015, OSWER Technical Guide for Assessing and Mitigating the Vapor Intrusion Pathway from Subsurface Vapor Sources to Indoor Air: U.S. Environmental Protection Agency, 212 p., accessed June 2016, at *https://www.epa.gov/sites/production/files/2015-09/documents/oswer-vapor-intrusion-technical-guide-final.pdf*.

U.S. Environmental Protection Agency, 2017, Summary Report, Vapor Intrusion Assessment, Vienna Wells Site, Vienna, Maries County, Missouri: U.S. Environmental Protection Agency, 12 p.

U.S. Geological Survey, 2012, National Water Information System Data available on the World Wide Web (USGS Water Data for the Nation), accessed May 8, 2012, at

https://waterwatch.usgs.gov/index.php?ym=201205&m=nwc.

Wilde, F.D., and Radtke, D.B., eds., 1998, National field manual for the collection of water-quality data—Field measurements: U.S. Geological Survey Techniques of Water-Resources Investigations, book 9, chap. A6, 238 p.

Wisconsin Department of Natural Resources, 1996, Analytical detection limit guidance and laboratory guide for determining method detection limits: Laboratory Certification Program, 33 p., accessed January 2014, at *http://dnr.wi.gov/regulations/labcert/documents/guidance/-lodguide.pdf*.