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Remedial Investigation Report, Thomson Consumer Electronics Facility, Circleville, Ohio

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Acronyms and Abbreviations

ASTM	American Society for Testing and Materials
ALM	adult lead model
BBL	Blasland Bouck & Lee
BBLES	Blasland Bouck & Lee Environmental Services
bgs	below ground surface
BKSF	Biokinetic Slope Factor
CCSEM	computer controlled scanning electron microscopy
CDC	Centers for Disease Control
CDI	chronic daily intake
CEC	cation exchange capacity
cfs	cubic feet per second
CLP	Contract Laboratory Program
CoI	chemical of interest
Consent Order	Administrative Order on Consent
cPAH	carcinogenic polycyclic aromatic hydrocarbon
CSF	carcinogenic slope factor
CSM	conceptual site model
DQO	data quality objective
DUR	Declaration of Use Restriction
EFA	East Fenced Area
EHWD	Earnhart Hill Water District
EPA	U.S. Environmental Protection Agency
EPC	exposure point concentration
EPtox	extraction procedure toxicity characteristic
FAQ	frequently asked questions
FSP	field sampling plan
GC/MS	gas chromatography/mass spectrometry
GE	General Electric Company
GFAA	graphite furnace atomic absorption spectrometry
gpd	gallons per day
gpm	gallons per minute
GSD	geometric standard deviation
HHRA	human health risk assessment
ICP-AES	inductively coupled plasma-atomic emission spectrometry
IEUBK	Integrated Exposure Uptake Biokinetic
IRIS	Integrated Risk Information System
LCS	laboratory control sample
LOAEL	lowest-observed-adverse-effect level
MCL	maximum contaminant level
NHANES	National Health and Nutrition Examination Study
NOAEL	no-observed-adverse-effect level
NPDES	National Pollutant Discharge Elimination System
NPDWR	national primary drinking water regulation

NTU	nephelometric turbidity unit
OCA	Offsite Creek Area
ODNR	Ohio Department of Natural Resources
OEPA	Ohio Environmental Protection Agency
OSHA	Occupational Safety and Health Administration
PAH	polycyclic aromatic hydrocarbon
Patridge	Patridge & Associates, Inc.
PCB	polychlorinated biphenyl
PCoI	potential chemical of interest
POTW	publicly owned treatment works
QAPP	quality assurance project plan
QA/QC	quality assurance and quality control
RAF	relative absorption factor
RBC	risk-based concentration
RCA	Radio Corporation of America
RCRA	Resource Conservation and Recovery Act
RfD	reference dose
RI/FS	remedial investigation and feasibility study
RME	reasonable maximum exposure
RPE	relative potency estimate
SDG	sample digestion group
SOP	standard operating procedure
SVOC	semivolatile organic compound
TAL	target analyte list
TCLP	toxicity characteristic leaching procedure
the Site	Thomson Consumer Electronics Circleville, Ohio site
Thomson	Thomson Consumer Electronics, Inc.
TOC	total organic carbon
TPH	total petroleum hydrocarbon
TSS	total suspended solids
UCL	upper confidence limit
USDA	U.S. Department of Agriculture
USGS	U.S. Geological Survey
VOC	volatile organic compound
WWTP	wastewater treatment plant

1 Introduction

A remedial investigation was conducted to satisfy the requirements of Attachment A Task 5, Section F, to the January 19, 1994 (Journalized on February 14, 1994) Administrative Order on Consent (Consent Order) with the Ohio Environmental Protection Agency (OEPA) for conducting a remedial investigation and feasibility study (RI/FS) at the Thomson Consumer Electronics Circleville, Ohio site (the Site). The remedial investigation activities were completed, following extensive historical investigations, in accordance with the RI/FS work plan (PTI 1995), as amended (GE 1997; BBL 2002, 2003, 2005, Exponent 2007) and approved by OEPA. Field investigation sampling was conducted by Blasland Bouck & Lee (BBL) for the sampling events performed prior to 2007. The 2007 sampling event was performed by ARCADIS BBL (formerly BBL). Data compilation and report preparation were conducted by Exponent (known as PTI Environmental Services prior to March 9, 1998), as supplemented by ARCADIS BBL.

A draft remedial investigation report was submitted to OEPA in April 1998. Following the draft remedial investigation report submittal, General Electric Company (GE) and OEPA discussed and reached agreement on the procedure necessary for finalizing the remedial investigation at the Site. In 2002, GE coordinated with OEPA to conduct an interim action involving limited soil/sediment removal along the west side of the railroad tracks at the Offsite Creek Area (OCA) to accommodate construction of a local access roadway by an offsite property owner. In 2003, GE also implemented supplemental soil/sediment sampling at the OCA and portions of the South Ditch to enhance delineation of lead in these areas. The supplemental data were collected to support remedial investigation data analysis and facilitate remedial decision-making for areas affected by offsite transport of lead-bearing particulates from the South Ditch. In 2005, GE performed supplemental sampling activities within portions of the OCA and South Ditch for the purpose of collecting sediment samples for analysis of total petroleum hydrocarbons (TPH). Finally, in 2007, GE conducted further supplemental sediment sampling at the OCA and portions of the South Ditch to collect sediment samples for biotoxicity testing as a component of the ecological risk assessment being conducted as part of the remedial investigation.

This remedial investigation report documents the investigation findings and data analyses completed up to and including the 2007 supplemental sampling event. Portions of the analyses completed for submittal of the draft remedial investigation report (e.g., transport and fate analyses in Section 5) were retained in their original form, except for specific revisions agreed upon with OEPA because the supplemental data did not change the earlier analyses and interpretations. The human health risk assessment (Section 6 and appendices) and Phase I ecological assessment (Section 7 and Appendix G) incorporate all data collected up to and including the 2007 supplemental sampling event. The purpose of this remedial investigation report is to document the investigation's entire findings and data analyses that will be used to facilitate the remedial decision making at the Site.

1.1 Site Description

The Site property consists of approximately 230 acres along the east side of U.S. Route 23, about 0.5 miles south of developed areas of the City of Circleville, Ohio (Figure 1-1). The plant Site is located approximately 700 ft above sea level on broad, gently sloping terraces of the Scioto River Valley. The Site is surrounded by open fields; a small residential area is located approximately 1,000 ft south-southwest of the plant area. The Scioto River is located approximately 0.75 miles west of the Site and is approximately 30–40 ft lower in elevation than the terraced uplands that include the plant.

1.2 Operations and Regulatory History

This section summarizes the Site manufacturing process and material usage, historical wastewater treatment, and solid and hazardous waste management practices at the Site. More detailed descriptions are provided in the work plan (PTI 1995). A summary of regulatory activities and previous response actions is included in the Findings of Fact section of the Consent Order (OEPA 1994). Key Site features are identified in Figure 1-2.

The plant was built in 1969 on a "greenfield site" and began operation in 1970 under the Radio Corporation of America (RCA). The plant was operated by RCA until 1986, when it was acquired through a corporate merger with GE. GE owned the Site for approximately 1 year before the Site was acquired by Thomson Consumer Electronics, Inc. (Thomson) in 1987. Thomson has owned and operated the Site from 1987 to the present. Thomson ceased manufacturing operations on March 30, 2004 and subsequently sold a large portion of their glass manufacturing equipment. This equipment was then removed from the facility. Thomson began demolition of the melting and forming operations areas of the Site in November 2005, and completed the demolition activities in 2006. Thomson has provided OEPA with a Site re-development plan indicating that anticipated future uses for the developed portion of the Site will be consistent with the current zoning regulations (i.e., commercial/industrial). At the appropriate time, Thomson will work with the current property Owner to establish an environmental restrictive covenant on the property pursuant to the Ohio Uniform Environmental Covenants Act, Ohio Revised Code §5301.80-5301.92, which will include activity and use limitations restricting possible future residential uses of the prior industrialized portion of the property. The portion of the property subject to the covenant is depicted on Figure 1-2. As indicated thereon, the boundaries of the portion of the property subject to the covenant will be as follows: the northern boundary of this area will be the property line between the Site and the Wal-Mart property to the north; the eastern boundary will be a north-south line 50 ft east of a similar north-south line passing through the eastern-most point of the East Fenced Area (EFA); the southern boundary will be the waterline on the southern bank of the South Ditch; and the western boundary will be a north-south line 25 ft west of a similar north-south line passing through the western-most point of the former wastewater treatment plant (WWTP) area. Information regarding plant operations has been obtained from records provided by Thomson personnel.

1.2.1 Manufacturing Process

The plant manufactured the face plate or panel (3 percent lead) and funnel (24 percent lead) components of television picture tubes from the time operations began in 1970 until shutdown in 2004. Prior to demolition activities, the plant consisted primarily of interconnected administration, production, laboratory, batch house, and warehouse buildings. Batch house silos were used to contain raw and intermediate materials such as sand, litharge (lead oxide), and cullet (recycled glass). Excess cullet was also stored on covered and uncovered concrete storage pads (Figure 1-3).

The following materials were used as glass components:

Cullet (recycled glass)	Barium carbonate
Silicon dioxide (sand)	Sodium antimonate
Sodium carbonate (soda ash)	Titanium oxide (rutile)
Potassium carbonate (potash)	Cobalt oxide
Strontium carbonate	Nickel oxide
Lead oxide (litharge)	Potassium dichromate
Sodium aluminum silicate (nepheline syenite)	Potassium silica fluoride (not used after 1975)
Potassium nitrate	Silica flour (no longer used)
Sodium nitrate	Arsenic trioxide (not used after 1986)

In addition, the following materials were used in the manufacturing process:

- Chrome plating solution used in mold preparation
- Cerium oxide, garnet, and pumice used as polishing abrasives
- Hydrofluoric acid/sulfuric acid used as polishing agents until approximately 1981
- Sodium *m*-bisulfite for treating chrome plating solution
- Lime for neutralizing chrome plating solution (following treatment) and acid waste
- Hydraulic oil and minor amounts of water-soluble oil used in process equipment
- Small volumes of solvents used in a parts washer in the maintenance shop.

The materials and process chemicals formerly used in the manufacturing process were reviewed during preparation of the work plan to develop a list of potential chemicals of interest (PCols) for the RI/FS. This review identified nine materials or process chemicals that contained 40 CFR Part 261 Appendix VIII constituents. These materials or process chemicals, the periods that the materials or process chemicals were used in manufacturing, the purpose of use, and the approximate quantities used are identified in Table 1-1. As indicated in Table 1-1, approximately 98 percent (by weight) of the materials used that contain any Appendix VIII constituents consist of litharge and glass cullet containing lead.

The raw materials were transported to the Site by rail or truck and unloaded in the batch house unloading areas at the east end of the former plant (Figure 1-3). Major bulk raw materials were stored in silos; minor raw materials in bags or buckets were stored in the enclosed batch houses.

There were three glass furnaces at the former plant: "A" for panel glass, "B" for funnel glass, and "C," built in 1991, for larger glass panels. For each furnace, raw materials were weighed and mixed in a batch plant. Cullet was added to this mixture. The batch mixture was then melted in one of the three furnaces, refined, pressed into shape, annealed, and inspected. The melting and molding took place in the "hot end." In the "cold end," the funnels received diamond grinding for proper mating with the face panel, and the panels were polished with inert abrasives (cerium oxide, garnet, and pumice). The finished components were shipped to other Thomson facilities for final television picture tube assembly.

In 1995, Thomson allocated a section of the Site to Praxair, Inc. to produce oxygen for the Thomson glass furnaces. The Praxair oxygen generation plant was put into production in 1996. The plant was located on the Site northeast of the Thomson manufacturing facility (Figure 1-2), but has now been disassembled and completely removed. Also in 1995, a cold neck seal manufacturing building was added to more efficiently process glass funnels (Figure 1-2). In this building, the glass funnels received diamond grinding of the seal edge and reference pads for proper matching with glass panels. A glass neck was cold-sealed to the glass funnels to receive an electron gun at the tube manufacturing plants.

1.2.2 Wastewater Treatment

The following sections discuss key elements of wastewater management and treatment at the plant, during the time of its operation. Historical wastewater treatment is described in more detail in Chester (1977).

1.2.2.1 Hot-end Cooling Water

From 1970 to 1990, the cooling water from the hot-end process was discharged at a rate of approximately 432,000 gal per day (gpd) to the Oil Skimmer Pond prior to discharge through former Outfall 001 to the South Ditch, an unnamed tributary of the Scioto River (Figure 1-2). OEPA issued a National Pollutant Discharge Elimination System (NPDES) permit for the hot-end cooling water discharge in 1974.

Another discharge from the hot-end area included air compressor/dryer condensate blowoff (less than 1,440 gpd), air dryer cooling water (approximately 7,200 gpd), and cullet cooling pad drainage (approximately 4,320 gpd) (Chester 1977). This water discharged to a storm water drain, which discharged to the South Ditch at an outlet located approximately 400 ft downstream of the former storm water outlet from the East Swale (currently storm sewer outlet B, Figure 1-2).

From August 1990 until 2006, all hot-end discharge water was conveyed to the WWTP, which was upgraded in 1990. Discharge of the former WWTP effluent is discussed below.

1.2.2.2 Cold-end Rinsewater

The glass particles and abrasives generated by the grinding and polishing operations were carried with the rinsewater used to rinse the glass parts. Prior to 1982, this cold-end rinsewater was managed by settling out the solids in lagoons and discharging the clarified water to the Circleville publicly owned treatment works (POTW). The settling lagoons were closed in the early 1980s as the WWTP was constructed and brought into service. From 1982 to 1990, the WWTP utilized two rubber-lined holding lagoons for settling of the clarified wastewater stream prior to final filtration and storage of excess filtered water for use in production operations. From 1982 to 1991, the cold-end rinsewater was treated in the WWTP prior to discharge to the POTW. From the time the WWTP was upgraded in 1990 until 2006, the treated cold-end effluent, combined with the hot-end effluent, was discharged at a rate of approximately 1 million gpd under a revised NPDES permit (Permit No. OEPA 4IN00030*HD) to the relocated Outfall 001 located approximately 60 ft upstream from the former Outfall 001 (Figure 1-2). Beginning in March 1997, non-contact cooling water was discharged to the South Ditch separately through Outfall 004 in compliance with NPDES permit revisions.

1.2.2.3 Storm Water

Prior to 1990, storm water that was collected in the storm sewers discharged into the South Ditch at three locations: the East Swale outlet, a storm drain outlet approximately 400 ft downstream of the East Swale, and a former storm drain outlet approximately 30 ft downstream from former Outfall 001 (Figure 1-2). Some storm water runoff from the cullet storage area may also have entered the East Swale. From 1990 until 2006, storm water runoff that came into contact with process materials at the east end of the "A/B" manufacturing plant, the north side of the "A/B" maintenance shop, or the "C" batchhouse material handling area, and all other runoff that collected in the East Swale (including Praxair storm water runoff) was diverted to a 200,000-gal storm water collection tank and then conveyed to the WWTP for treatment consisting of filtration prior to discharge through a permitted outfall.

1.2.2.4 Chrome Plating Rinsewater

Until 1991, the chrome plating rinsewater was treated by addition of sodium *m*-bisulfite to reduce hexavalent chromium (Cr[VI]) to trivalent chromium (Cr[III]) and precipitated with lime. The treated effluent was historically discharged to Lagoons 1 and 2 and then to the Circleville POTW sanitary sewer. The treated effluent was conveyed directly to the WWTP

when it went into operation in 1982. From 1991 until 2006, plating rinsewater was recycled without treatment in the new chrome plating system.

1.2.2.5 Water Supply Treatment

Water for plant processes and for drinking was supplied solely by the City of Circleville until 1985, when two groundwater production wells and a small water treatment plant were installed in the southeast portion of the Site (Figure 1-2). Water treatment consisted of chlorination and a sand filtering process to remove iron and manganese. Two outfalls, 002 and 003, were used under an NPDES permit to discharge filter backwash from the water supply treatment plant to the South Ditch. Outfalls 002 and 003 were permitted for discharge at an estimated combined flow rate of approximately 2,650 gpd.

1.2.3 Solid and Hazardous Waste Management

This section describes how solid and hazardous wastes were generated and managed during the plant's operation.

1.2.3.1 Glass Fines

Prior to 1980, glass polishing and grinding materials (fines) were carried in the cold-end rinsewater and settled out in two lagoons (1 and 2) that were located in the area between the offices and what was the "C" manufacturing building. The solids were pumped out of the lagoons approximately four times per year to sludge pits in the EFA. The OEPA preliminary assessment report (OEPA 1987), referring to April 1980 inspection files, indicated that three pits were approximately 2 acres in area and were up to 8 or 9 ft deep. The pits were covered with 1–2 ft of clean soil in October 1980.

Sludge that accumulated between 1980 and 1982 in Lagoons 1 and 2 (discussed above) and in Lagoons 3 and 4 (temporary impoundments used in 1981 and 1982 during construction of the WWTP that were located just east of Lagoons 1 and 2) was disposed of offsite at an appropriate landfill after testing for lead by the extraction procedure toxicity characteristic (EPtox). Sludge testing positive (i.e., with soluble lead concentrations higher than 5.0 mg/L) was transported to a permitted Resource Conservation and Recovery Act (RCRA) landfill (Unico Landfill, Plain City, Ohio). Otherwise, the sludge was disposed of at a commercial solid waste landfill (Pine Grove Landfill, Amanda, Ohio).

From 1982 until 2006, wastewater containing abrasives and glass fines was routed through the WWTP; the sludge was collected as filter cake from the vacuum filter of the WWTP and disposed of offsite. The sludge was tested routinely by EPtox analysis for arsenic, chromium, and lead and also by the toxicity characteristic leaching procedure (TCLP) for lead and chromium. When the EPtox or TCLP extract limits for lead were exceeded, the sludge was disposed of at a RCRA-permitted hazardous waste landfill (Waste Management, Fort Wayne, Indiana). However, exceedances of the TCLP extract limit for lead were virtually eliminated following the improvements made to the WWTP in 1990.

1.2.3.2 Hazardous Wastes

The following wastes generated at the facility were historically designated as hazardous (U.S. Environmental Protection Agency [EPA] waste codes D005, D007, and D008) on the basis of metals concentrations in the EPtox or TCLP analysis: floor sweepings and baghouse dust from the batch plants, furnace checker particulates and electrostatic precipitator dust from the furnaces, and, occasionally, batches of glass fines from the funnel grinding operation. These wastes were accumulated onsite prior to offsite disposal at a permitted facility. The waste materials were consistent throughout the operation of the facility. During the 1970s, D005, D007, and D008 wastes were temporarily accumulated in a pile at the location of the former hazardous waste storage building. During this period, the waste was piled either on the soil surface or on a concrete pad. After the hazardous waste storage building was constructed in 1980, all D005, D007, and D008 wastes were contained in it prior to offsite disposal. From the early 1980s until 2006, electrostatic precipitator dust, and, from the late 1980s until 2006, funnel glass fines, were recycled at the plant. On June 22, 1992, EPA approved clean closure of the hazardous waste storage building in accordance with 40 CFR Part 264; a summary of soil sampling for this area is available in the clean closure equivalency demonstration report (PTI 1992). Until 2006, floor sweepings, baghouse dust, and furnace checker particulates were accumulated for less than 90 days in containers in the hazardous waste storage building (Figure 1-3) before disposal offsite at RCRA-permitted facilities.

The spent mold plating baths and precipitates from the treatment of chrome plating rinsewater were also classified as hazardous (EPA Waste Codes D002 and D007). These wastes were also contained, accumulated in the hazardous waste storage building, and disposed of offsite in accordance with applicable regulations.

1.3 Previous Investigations

The Site has been the subject of a number of environmental studies and investigations since 1988. More than 200 groundwater, 9 surface water, 452 soil/sediment, and 47 biological tissue samples have been collected and analyzed prior to the remedial investigation activities, which began in late 1995. Previous investigations at the Site are discussed in detail in the work plan (PTI 1995). Where appropriate, historical data were incorporated in the scoping of the remedial investigation activities and in the preparation of this remedial investigation report.

1.4 Interim/Emergency Actions

This section summarizes interim or emergency actions that have been conducted at the Site to improve environmental conditions and facilitate more efficient plant operations. These actions are consistent with RI/FS objectives for the Site.

1.4.1 Clean Closure of Former Settling Lagoons

The former settling lagoons previously used for management of glass polishing fines (Lagoons #1 and #2, then #3 and #4), were clean closed under RCRA Interim Status (40 CFR

Part 265) in 1982 and 1984, respectively (Keystone 1990). In response to an OEPA site screening inspection, soils in the vicinity of the former settling lagoons were removed in 1988 and disposed of offsite at an approved disposal facility. These lagoons subsequently received clean-closure approval by EPA under the more stringent 40 CFR Part 264 requirements in July 1990.

Following soil removal activities, confirmation soil sampling during closure demonstrated that remaining soils did not contain applicable Appendix VIII constituents at significant levels. The clean closure mitigated any future potential release of PCoIs from the vicinity of the lagoons, and likely contributed to the decreasing trend in fluoride concentrations observed in shallow groundwater at the Site.

1.4.2 Clean Closure of Hazardous Waste Storage Building Unit

The former hazardous waste storage building was used as a 90-day accumulation area for hazardous wastes generated from glass manufacturing (Figure 1-3). The unit was clean closed under RCRA Interim Status (40 CFR Part 265) in 1985. Subsequent to an OEPA site screening inspection, soils in the vicinity of the hazardous waste storage building unit were removed in 1988 and disposed of offsite at an approved disposal facility. The unit subsequently received clean-closure approval by EPA under the more stringent 40 CFR Part 264 requirements in 1992 (PTI 1992). The building was routinely inspected and managed in accordance with applicable regulations for the accumulation of hazardous waste.

1.4.3 Soil Cover and Fencing at the EFA

Placement of glass polishing fines (sludge) at the EFA was discontinued in 1980. In 1980, the sludges were covered with approximately 2 ft of clean fill. The EFA was also enclosed with a perimeter fence in 1989. These measures help control any access and possible exposure to the sludge deposits. The soil cover also provides protection from potential dispersal by wind action.

1.4.4 Upgrade of the Wastewater Treatment System

The capabilities for onsite wastewater treatment were expanded over the period of plant operation through multiple upgrades and modifications. Construction of the WWTP in 1982 eliminated discharge of the cold-end effluent to the unlined lagoons. Significant WWTP upgrades in 1990 allowed for the onsite treatment of hot-end effluent generated at the plant and treatment of storm water from process areas of the plant. These efforts eliminated discharges of process water to the POTW, the need for the Oil Skimmer Pond, and the discharge of untreated storm water from the east end of the plant and the East Swale to the South Ditch. Wastewater treatment system upgrades also resulted in more efficient treatment of the effluent.

1.4.5 Storm Water Collection and Treatment System

Prior to 1990, storm water that was collected by the Site storm sewers discharged into the South Ditch at three outfalls (Figure 1-2). These three outfalls and corresponding drainage areas were as follows:

- The East Swale Outlet: agricultural and indeterminate surface areas to the east of the plant buildings
- A storm drain outlet approximately 400 ft downstream of the East Swale (referred to as 19+30 outfall, storm sewer outlet B): cullet storage, north-side employee parking lots, cafeteria and maintenance shop roofs, railroad spur loading/unloading areas, and a portion of "A/B" manufacturing roof
- A former storm water drain outlet approximately 30 ft downstream of former NPDES Outfall 001: main office roof, warehouse roof, southwest employee and visitor parking lots, and the area around the WWTP.

From 1990 until 2006, storm water runoff that came into contact with process materials at the east end of the "A/B" manufacturing plant, the north side of the "A/B" maintenance shop, "C" batchhouse material handling area, and all other runoff that collected in the East Swale (including Praxair storm water runoff) was diverted into a "lift station" and pumped to a 200,000-gal storm water collection tank. The storm water collection tank was metered into the WWTP for treatment, consisting of filtration, prior to discharge through an NPDES-permitted outfall.

1.4.6 Removal of Oil Skimmer Pond

The Oil Skimmer Pond was removed from service in 1990. In 1992, soils and sludge were excavated from the bottom and sides of the pond to approximately 10 ft below ground surface (bgs) and were disposed of offsite as part of a removal action. Excavation of the soils around the pond removed a potential source of PCoIs to media in the immediate area.

1.4.7 Diesel Spill Response

Approximately 300 to 800 gal of diesel fuel were spilled on the ground at the "C" manufacturing building on November 30, 1991. Corrective actions resulted in the recovery of approximately 70 gal of fuel and 310 gal of fuel/water mixture from shallow soils adjacent to the building. Phase I and II investigations were subsequently implemented and coordinated with OEPA; shallow groundwater wells (MW-10 through MW-13) were installed and soil and groundwater samples were collected and analyzed for fuel constituents. Sampling and analysis indicated that the environmental impact was negligible and OEPA required no further investigation or remediation (OEPA 1993).

1.4.8 Offsite Creek Interim Action

In September 2002, an interim action to remove lead-bearing soil/sediment at the OCA to accommodate construction of a local access roadway by the current offsite property owner, Richards Land Company (Richards) was implemented. Soil and sediment within the footprint of the roadway with lead concentrations exceeding 400 mg/kg were removed and disposed of offsite in accordance with applicable regulations. The 400 mg/kg lead concentration value was specified in the OEPA-approved *Interim Action Work Plan for Select Soil/Sediment Removal from the Offsite Creek Area* (BBL 2002). As indicated therein, the basis for such removal was OEPA's Residential Land Use Standard of 400 ppm. The interim action included construction of a culvert extension (approximately 75 ft in length), and removal of 1,559 tons (approximately 1,039 yd^3) of soil/sediment immediately west of the railroad tracks. A work plan detailing the interim action approach was submitted to OEPA on September 5, 2002, and revised on September 27, 2002, in response to OEPA comments (BBL 2002). The final work plan was approved by OEPA in letters to GE dated October 4 and 10, 2002. A final report documenting the completion of the interim action was submitted to OEPA on February 11, 2003 (BBLES 2003).

1.5 Potential Chemicals of Interest

PCoIs were identified during the remedial investigation scoping process by examining known or assumed compositions of past raw materials and process chemicals, and by reviewing results of all known previous investigations. A complete description of the evaluation process is provided in the work plan (PTI 1995). Based on this evaluation, the following chemicals were considered as PCoIs for the remedial investigation:

- Antimony
- Arsenic
- Barium
- Chromium
- Lead
- Nickel
- Fluoride
- TPH.

During the remedial investigation, polycyclic aromatic hydrocarbons (PAHs) were detected at elevated concentrations in sediment samples collected from portions of the South Ditch; therefore, PAHs were also considered PCoIs.

1.6 Areas of Interest

Information regarding historical plant operations, waste management practices, Site setting, and results of previous investigations were reviewed to identify areas of interest to be evaluated during the remedial investigation. Based on this review, the following areas were identified for further investigation and are shown on Figure 1-4:

- East Fenced Area (EFA)
- Adjacent Fields
- East Swale
- Former Oil Skimmer Pond
- South Ditch
- OCA

In addition, the work plan evaluated another area referred to as Onsite Soils. Review of the historical data (PTI 1995) indicated that the Onsite Soils area had negligible concentrations of PCoIs, and this area was excluded from further field investigations; however, the data for the Onsite Soils were incorporated in the screening analyses of the human health risk assessment (Chapter 6), which confirmed the interpretations presented in the work plan (PTI 1995).

2 Field and Laboratory Investigations

This section briefly describes, by area of interest, the field and laboratory investigations that were conducted as part of the remedial investigation. More detailed background descriptions of these investigations can be found in the work plan and its appendices (PTI 1995), as amended (GE 1997; BBL 2002, 2003, 2005; Exponent 2007) and approved by OEPA. The prefixes used for the names of specific sampling stations in this and subsequent sections are defined in the field sampling plan (FSP) (Appendix A of PTI [1995]). For each area of interest at the Site, subsections of this report describe the objectives for the collection of data and the approach to field sampling and laboratory analyses. Deviations from the work plan (PTI 1995), as amended, are described in Section 2.9. Results from these investigations are presented in Section 4, *Nature and Extent of Contamination.*

Although lead was the primary target chemical for all areas of interest in the remedial investigation, full PCoI metals analyses were conducted on approximately 25 percent of all soil and sediment samples collected. The work plan (PTI 1995) describes the specific procedures followed to determine the additional analyses requested on archived samples at the contract laboratory during the remedial investigation.

2.1 East Fenced Area

The EFA comprises approximately 5 acres located east of the former plant (Figure 1-4), and was the historical location for disposal of sludge from the facility. Glass polishing and grinding fines were pumped from lagoons to the sludge pits in the EFA during the 1970s. Data from previous soil and sludge sampling at the EFA are discussed in detail in the work plan (PTI 1995).

2.1.1 Objectives

The objectives of the field activities conducted at the EFA were to:

- Determine the horizontal extent and permeability of the sludge
- Characterize and delineate the extent of soils with elevated lead concentrations at former Dames & Moore sample location 25D located to the west of the EFA fence (Figure 2-1)
- Confirm the hydraulic separation of the shallow groundwater zone and underlying deep aquifer
- Establish an understanding of the seasonal groundwater flow system in the shallow groundwater zone
- Evaluate the presence or absence of PCoIs in groundwater at the EFA.

2.1.2 Approach

2.1.2.1 Backhoe Excavations, Confirmation Soil Sampling, and Sludge Permeability Testing

The lateral extent of sludge was determined with shallow (approximately 2 ft deep) backhoe scrapings within the fenced area, west of the fence perimeter (in the vicinity of former sampling station 25D), and east-southeast of the fence perimeter. Backhoe excavations were conducted at 27 locations on a grid with approximately 100-ft centers (Figure 2-1). Additional hand-dug pits were used to further define the extent of sludge to the east and southeast of the fenceline. The sludge is visibly distinct and easily distinguished from native soils. The sludge boundary was confirmed through collection of soil samples from four locations beyond the perimeter of the visually identified sludge placement area. All four samples were analyzed for lead, and two samples were analyzed for additional PCoIs.

To assess the vertical permeability of the sludge to infiltrating rainwater, two percolation tests were conducted. Following delineation of sludge boundaries, a sampling station was selected in the middle of the largest sludge deposit area. A backhoe was used to scrape away the surface soil cap and expose a fresh sludge surface to conduct an *in situ* percolation test, which was performed twice. Any disturbed sludge and overlying soils were placed back into the excavation following completion of the percolation tests.

2.1.2.2 Soil Sampling At Station 25D

An elevated lead concentration had been reported by Dames & Moore (1991) at Station 25D, located to the west of the EFA fenceline. The elevated lead concentration was similar to sludge sample concentrations, and its location next to the EFA indicated that the elevated lead was likely associated with historical sludge management activities at the EFA. Because the extent of elevated lead at Station 25D could not be visually delineated (as sludge), confirmation soil samples were collected at the four cardinal positions around former Station 25D. A sample was collected at each cardinal position from the 0-to-6-in. interval, and from the 0-to-6-, 6-to-12-, and 12-to-18-in. intervals from the middle of the area believed to be affected.

2.1.2.3 Borehole Drilling

Four boreholes were advanced around the perimeter of the EFA (Figure 2-1) to collect soil samples to describe stratigraphy and evaluate the presence and depth of the low-permeability clay unit, which had been estimated to be present at a depth of 30–35 ft bgs. Borehole locations were determined in the field based on the backhoe scrapings and visual observations to confirm that boreholes were placed just beyond the perimeter of the sludge deposits. Three of the four boreholes were overdrilled for stratigraphic description and were ultimately used to construct monitoring wells (MW-14, MW-15, and MW-16); the fourth borehole was installed immediately east of the EFA to confirm stratigraphic conditions in this direction.

All boreholes were advanced using an 8-in.-diameter hollow-stem auger to maximum depths from 27 to 54 ft bgs. Boreholes BH-16 and BH-EFA-1 were advanced into the top of clay deposits observed. A sample of clay deposits observed at 27 ft bgs in BH-EFA-1 was collected

for determination of physical properties. Each of the three boreholes that were used to install monitoring wells (BH-14, BH-15, and BH-16) was backfilled (sealed) below the target depth prior to well construction.

Continuous soil sampling was conducted during drilling at all borehole locations and samples were described and logged on field forms using American Society for Testing and Materials (ASTM) methods. All soil samples were archived for possible additional testing and analysis. A clay layer sample was analyzed for physical parameters including bulk density, moisture content, particle size distribution, Atterburg limits, porosity, and hydraulic conductivity. Samples were collected with split-spoon or thin-wall tube-type (Shelby®-type) samplers. Borehole logs are located in Appendix A.

2.1.2.4 Monitoring Well Construction and Development

Three of the boreholes advanced at the EFA (Section 2.1.2.3) were completed as monitoring wells (MW-14, -15, and -16). Each well was completed such that the 10-ft screened interval was expected to intercept the water table throughout the year. The fourth borehole located east of the EFA was abandoned in accordance with the work plan.

Monitoring wells were constructed using a 2-in.-diameter casing/screen assembly. Well casings were constructed of Schedule 40 PVC with flush-threaded joints, a threaded 0.5-ft-long sump with end cap, and a 10-ft-long Schedule 40 PVC well screen with machine cut slots (0.020-in. slot size). Because of problems associated with well construction after overdrilling, MW-16 was later abandoned and a replacement well (MW-16R) was installed nearby using the same construction techniques and well materials, except that a 0.010-in. slot size well screen was used to help reduce turbidity. Centering guides were used at the base and top of the screens and a locking end cap was installed on each well. Specific methods and procedures followed for the borehole and monitoring well completion are described in the FSP (PTI 1995). Monitoring well and borehole locations at the EFA are shown on Figure 2-1. Well construction details are summarized in Table 2-1.

Following monitoring well completion, the wells were developed by surging with a stainless-steel bailer and purging with a Grundfos® submersible pump to remove fines and set the sand pack. Indicator parameters (pH, electrical conductivity, and temperature) were measured periodically during development. The wells were purged by removing a minimum of 25 wellbore volumes, or until the indicator parameters had stabilized and the turbidity was reduced as much as possible.

2.1.2.5 Groundwater Elevation Monitoring

During the remedial investigation, the three new monitoring wells at the EFA were incorporated into the Site-wide water elevation monitoring network to document the interaction of surface water and groundwater and to refine our understanding of groundwater flow directions and gradients (discussed further in Sections 3.4.2.2 and 3.5.1). The Site-wide monitoring network (shown in Figure 2-2) consists of the existing Site monitoring wells, the new EFA wells, and piezometers and staff gauges installed along or in the vicinity of the South Ditch.

Revised reference elevations were established for all existing wells during topographic mapping at the Site. All wells, piezometers, and staff gauges installed during the remedial investigation were also surveyed relative to the Site datum. Site-wide groundwater elevation data were collected monthly for 1 year to evaluate seasonal and spatial variations in groundwater flow direction and gradient.

2.1.2.6 Groundwater Sampling

Following well installation and development, monitoring wells MW-14, MW-15, and MW-16 were sampled for total and dissolved PCoI metals and total fluoride on a quarterly basis for five quarters using a Grundfos® submersible or peristaltic pump.

2.2 Adjacent Fields

The Adjacent Fields area is located immediately north of the facility (Figure 1-4). Previous sampling events conducted north of the Site boundary are summarized in the work plan.

2.2.1 Objectives

The objectives of the Adjacent Fields sampling were to:

- Collect data to confirm fluoride, lead, and other PCoI concentrations in soil immediately downwind of the manufacturing plant
- Demonstrate that potential historical air emissions did not result in PCoI concentrations in Adjacent Fields soils at levels of concern.

2.2.2 Approach

Four field sampling locations were established along and immediately south of the property boundary immediately north of the "A" and "B" manufacturing buildings (Figure 2-3). These sampling locations were based on the historical predominant wind direction (from the south), the preference for sampling undisturbed soil (lack of soil tilling on this portion of the Site), and consideration of previous OEPA (1980) sampling stations in the plowed farm fields immediately north of this area.

Soil samples were collected from two depth intervals at each location (0-to-6-in. and 6-to-12-in. depths). The surface of each sample location was carefully cleared of vegetation prior to sampling. One of the 0-to-6-in.-interval samples was analyzed for all seven PCoIs and the other three were analyzed only for lead and fluoride. Two of the four samples collected from the 6-to-12-in. interval were analyzed for all seven PCoIs and two were analyzed only for lead and fluoride. In addition, one sample was analyzed for pH, cation exchange capacity (CEC), and total organic carbon (TOC).

2.3 East Swale

The East Swale is the drainage ditch located east of the plant (Figure 1-4). The East Swale is typically dry; however, during significant precipitation events, the East Swale may receive runoff from fields northeast of the plant and a portion of the storm water runoff from the area at the east end of the plant that was formerly used to store cullet and raw materials. As discussed in the work plan, a limited number of samples were previously collected from the East Swale.

2.3.1 Objectives

The objectives of the sampling conducted at the East Swale were to:

- Complete the delineation of the extent of elevated lead in sediment and soil and confirm the relationship of lead to other PCoIs
- Analyze specific PCoIs (antimony, nickel, and fluoride) in sediment and soil samples that have not been analyzed in previous investigations at the swale to confirm that lead is the target chemical
- Characterize the mineralogy of lead and percent glass fraction
- Confirm the capacity of sediment/soil to immobilize metals.

2.3.2 Approach

To augment data previously collected by Dames & Moore (1989), four transects were established at approximately even intervals along the length of the swale (Figure 2-4). The locations of transects allowed for the delineation of lead concentrations in sediment and soil and provided data to evaluate concentrations of other PCoIs not previously analyzed in this area. Similar to the previous sampling conducted by Dames & Moore, the transects were established perpendicular to the ditch axis with sampling locations and depths as follows:

- Location A: surface sediments in central channel; 0-to-6-in. depth interval
- Location B: core sample below the channel sediments; 0-to-6-, 6-to-12-, 12-to-18-, 18-to-24-, and 24-to-30-in. depth intervals below Location A
- Location C1: core samples on either side of the channel (right and left facing upstream) at the sediment contact with the base of the swale slope; 0-to-6-, 6-to-12-, and 12-to-18-in.-depth intervals
- Location C2: core samples on either side of the channel (right and left facing upstream) near the first break-in-slope out of the channel; 0-to-6-, 6-to-12-, and 12-to-18-in.-depth intervals.

All samples were analyzed for lead, and selected samples were analyzed for other PCoIs. In addition, samples collected from each transect at the "A" 0-to-6-in. depth interval were analyzed for pH, TOC, CEC, lead mineralogy, and percent glass fraction to evaluate the attenuation of metals.

2.4 Former Oil Skimmer Pond

The former Oil Skimmer Pond was located south of the main plant area (Figure 1-4). The Oil Skimmer Pond was used to remove oil from hot-end cooling water until 1990. The Oil Skimmer Pond was removed in 1992. As discussed in the work plan (PTI 1995), limited sampling was conducted in the area during excavation of the soils around the pond.

2.4.1 Objectives

The objectives of the field activities at the former Oil Skimmer Pond were to:

- Conduct confirmation sampling for lead and TPH in soil beneath the pond backfill
- Evaluate the reported oily sand horizon east of the former pond and collect confirmation soil samples for TPH analysis
- Analyze hydraulic-oil-containing soil for organic compounds to confirm that hazardous constituents are not present at significant concentrations.

2.4.2 Approach

Two boreholes (BHOSP-01 and -02) were drilled through the former pond backfill materials (Figure 2-5). Soil samples were collected from each borehole from immediately beneath the fill. Both samples were analyzed for PCoI metals, TPH, volatile organic compounds (VOCs), and semivolatile organic compounds (SVOCs).

To evaluate the extent of the reported oily soil horizon, test pits were excavated at five locations east of the former pond (Figure 2-5). Pits were excavated to depths between 4 and 6 ft. Soil samples were collected from each pit and analyzed for TPH and for VOCs and SVOCs in one representative sample. In addition, to supplement test pit evaluation, subsurface soil samples were collected from six additional locations using Geoprobe® sampling methods (Figure 2-5). Two Geoprobe® samples (CN-OSP-6 from Station H and CN-OSP-7 from Station J) were sent to an analytical laboratory for TPH analysis. Soil samples from the other four Geoprobe® locations were analyzed for VOCs using a field-based gas chromatograph.

2.5 South Ditch

The South Ditch is the onsite portion of the unnamed tributary to the Scioto River that is located on the south side of the plant (Figure 1-4). Multiple current or historical outfalls are located on the ditch and, prior to 1990, the East Swale drained into the ditch during significant storm water runoff events (Figure 1-2). Also, prior to 1992, the former Oil Skimmer Pond discharged to the South Ditch. Extensive surface water, soil, and sediment sampling has been conducted historically along the South Ditch and is described in detail in the work plan (PTI 1995).

2.5.1 Objectives

The objectives of the remedial investigation activities at the South Ditch were to:

- Delineate the vertical extent of lead in sediment at Dames & Moore (1989) transects 0+25 and 19+30, and analyze for all PCoI metals in representative samples
- Confirm the capacity of sediments to immobilize lead
- Conduct lead mineralogy and percent glass fraction analysis to confirm estimates of geochemical fate and chemical stability
- Evaluate the seasonal hydraulic relationship of ditch water levels to the shallow water table zone and confirm representativeness of historical groundwater quality data
- Sample sediment and surface water near the east end of the Site boundary to characterize upstream conditions
- Analyze oily sediments for VOCs, SVOCs, and polychlorinated biphenyls (PCBs) to confirm that hazardous constituents are not present at significant concentrations.

Supplemental remedial investigation sampling was conducted in 2003 to further delineate the extent of lead along the South Ditch west of the former East Swale outfall to the ditch to support remedial decision-making. Additional sampling was also performed in 2005 to determine the presence and extent of TPH within South Ditch sediments. Finally, supplemental sampling was performed in 2007 in support of sediment bioassays and revisions to the ecological risk assessment.

2.5.2 Approach

The approach used to meet the remedial investigation objectives is presented in the following sections. Historical and remedial investigation South Ditch sampling locations are presented in Figure 2-6.

2.5.2.1 Sediment/Soil Sampling

The transect sampling conducted by Dames & Moore (1989) demonstrated that lead is strongly attenuated in the surficial sediments and soils in the South Ditch with elevated lead concentrations generally restricted to the upper 6–12 in. However, the vertical extent of elevated lead was not completely delineated at two transect sample locations. The "C-North" location at station 0+25 and "B" location at station 19+30 had elevated lead concentrations at the base of the sampled cores (12 and 18 in., respectively).

To complete delineation of the extent of lead in the South Ditch, the vertical extent of elevated lead concentrations was determined at these two former sampling locations. Core samples were collected to a total depth of 30 in. and analyzed for lead at 6-in. intervals. Two sediment samples (i.e., one from each location) were analyzed for all PCoI metals to confirm the previously observed relationship between lead and the other metals.

Samples at three locations along the ditch were collected to characterize lead mineralogy and the capacity of the sediment to immobilize lead. Sandy deposits were collected at former Dames & Moore sample stations 0+25, 13+60, and 21+05. These locations were chosen to represent sediment conditions that are typical of the South Ditch substrate. The samples used to evaluate immobilization capacity of the sediments were collected from the "B" transect location (i.e., beneath the central channel sediments) at the 0-to-6-in. depth interval. The samples were analyzed for pH, CEC, TOC, lead mineralogy, and percent glass fraction. Samples of the active channel sediment (Location A) were also collected at three locations and analyzed for grain size.

Following initial sediment sampling at two stations where PAHs were detected (G-SD-2 and G-SD-3), four additional samples were collected at stations 0+25 and 19+30 to further assess PAH concentrations (Figure 2-6). Sampling stations G-SD-2 and G-SD-3 were located near and slightly downstream from the former Oil Skimmer Pond outfall to the South Ditch to confirm that organic hazardous constituents are not present at significant concentrations in sediments containing hydraulic oil. These samples were collected from a depth of approximately 0–6 in. and analyzed for VOCs, SVOCs, and PCBs. Samples were collected from both the north and south banks at 0+25 and 19+30 and analyzed for SVOCs to assess PAH concentrations potentially derived from the storm water outfall at station 19+30 (storm sewer outlet B on Figure 1-2).

Following completion of the initially planned remedial investigation sampling, supplemental transect sampling was conducted in 2003 to further delineate the extent of lead along the South Ditch west of the former East Swale outfall. Soil/sediment samples were collected along each transect from ditch centerline, side of ditch, and top of bank locations up to a maximum total depth of 21 in. (depth of refusal) and analyzed for lead at 6-in. intervals. Some of the collected soil/sediment samples were archived at the laboratory and were analyzed for lead if the overlying or nearby sample indicated a significantly elevated lead concentration. The supplemental sampling locations are shown on Figure 2-6.

The supplemental TPH investigations were performed in 2005 to determine the presence and extent of TPH within the South Ditch sediments (BBL 2005). Sediment samples were collected from the 0- to 6-in., 6- to 12-in., and 12- to 24-in. depth increments at nine locations approximately evenly spaced along the South Ditch and analyzed for TPH. The samples from

the two deeper intervals were held at the laboratory and were analyzed for TPH if the overlying samples contained TPH at a concentration greater than the maximum allowable residual TPH concentration of 5,000 ppm for Type I soils, as defined in OEPA's Division of Emergency and Remedial Response's January 14, 2004, guidance document titled *Soil Leaching to Groundwater Evaluation for Total Petroleum Hydrocarbons* (OEPA 2004a). The TPH sampling locations are shown on Figure 2-6.

Finally, supplemental sediment sampling was performed in support of sediment bioassays and revisions to the ecological risk assessment (Exponent 2007). Sediment samples were collected from the 0- to 6-in. depth increment using an Ekman dredge. Several samples were collected from certain previous remedial investigation sampling locations in an attempt to collect samples representative of the following concentration ranges: 40 to 400 ppm, 400 to 800 ppm, and greater than 800 ppm. Samples were analyzed for lead, TOC, acid-volatile sulfide/semi-extractable metals (AVS/SEM), and grain size. Samples representative of the specified target concentration ranges were selected for subsequent sediment bioassay testing. The sediment sampling locations associated with the sediment bioassay testing are shown on Figure 2-6.

2.5.2.2 Upstream Sediment and Surface Water Sampling

To provide an understanding of conditions upstream of the South Ditch, sediment and surface water quality samples were collected from the ditch near the east end of the Site in an area unaffected by plant operations (Station G-SD-1, Figure 2-6).

One composite sediment sample was collected from the ditch and analyzed for all PCoIs. The composite consisted of three grab samples collected from the 0–6 in. depth in a single transect across the active portion of the ditch.

As a control for the TPH and sediment bioassay investigations, sediment samples were collected from the portion of the South Ditch upstream of the former plant site and (for the bioassay investigation) an adjacent farm ditch, to provide sediment that was representative of background conditions/unaffected by the former plant operations. These background sampling locations are presented on Figure 2-6.

Upstream surface water samples were collected at low and high (storm event) conditions and analyzed for total and dissolved lead. Ditch flow (discharge), total suspended solids (TSS), hardness, pH, conductivity, temperature, and dissolved oxygen were measured in the field.

2.5.2.3 Staff Gauge and Piezometer Installation

The seasonal hydraulic relationship of ditch water levels to the shallow groundwater table zone was evaluated with the installation of staff gauges and piezometers along the ditch. Water elevation data from these stations and all other Site monitoring wells were collected and used to characterize this relationship and better define the interaction of surface water and shallow groundwater.

Staff gauge and piezometer locations are provided on Figure 2-7. Six staff gauges were installed in the ditch to collect surface water levels. Where possible, the staff gauge and

piezometer were installed adjacent to monitoring wells. Five of the piezometers were installed near the staff gauges along the south side of the ditch to provide hydraulic control in this area. A sixth piezometer was installed next to MW-9 to evaluate the vertical gradient between the shallow water table zone and deep aquifer.

2.5.2.4 Water Elevation Monitoring

As mentioned previously, reference elevations for the piezometers, staff gauges, and monitoring wells were surveyed relative to the Site datum. Water level measurements were collected from the staff gauges, piezometers, and Site monitoring wells on an approximate monthly basis between December 1995 and April 1997, with an additional event in August 1997. Groundwater flow directions and the seasonal interaction of surface water and shallow groundwater were determined from the monitoring. The vertical gradient between the shallow water table zone and deep aquifer was also determined at MW-9.

2.6 Offsite Creek Area

The OCA is located west of the facility downstream of the South Ditch to the west of Highway 23 (Figure 1-4). The OCA refers to the narrow riparian corridor approximately 4,000 ft in length and located between Highway 23 and the Scioto River (approximately 12 acres in total area). This corridor is formed along two principal drainage segments referred to as the offsite creek and the Farm Ditch and immediate overbank areas west of the railroad tracks. The offsite creek is the extension of the South Ditch between the highway and the Farm Ditch that drains along the large agricultural field located on the floodplain of the Scioto River. The portion of the drainage between Highway 23 and the railroad tracks is currently referred to as the upper creek. Runoff from nearby residential/commercial and agricultural areas and effluent from the Earnhart Hill Water District (EHWD) water treatment plant also discharge to the OCA drainage system. As described in the work plan (PTI 1995), previous sampling indicated that lead was present at elevated levels in a limited portion of OCA soils/sediments (primarily within a marshy area herein referred to as the deltaic area). Additional descriptions of the OCA and surrounding area are provided throughout this report and in Appendix G.

2.6.1 Objectives

The objectives of the OCA investigation were as follows:

- Define the depositional processes and environments in the OCA
- Complete and confirm delineation of lead in channel sediments
- Confirm the relationship of lead to other PCoIs
- Confirm that lead-bearing particulates have not migrated downstream (e.g., from removed beaver dams) since previous investigation sampling and following limited ditch maintenance activities by Richards

- Complete delineation of the extent of lead in sediment in overbank deposits near the confluence of the offsite creek with the south-flowing farm ditch and analyze selected samples for all PCoIs to confirm the relationship between lead and the other PCoI metals found during previous sampling at the Site
- Characterize lead mineralogy and confirm the capacity of sediments to immobilize lead
- Evaluate lead concentrations in surface water.

Supplemental remedial investigation sampling was conducted in 2003 to further delineate the extent of lead at the OCA and to support remedial decision-making for lead-bearing particulates transported to the OCA from the South Ditch. Additional sampling was also performed in 2005 to determine the presence and extent of TPH within South Ditch sediments. Finally, supplemental sampling was performed in 2007 in support of sediment bioassays and revisions to the ecological risk assessment.

2.6.2 Approach

Activities conducted to meet the RI/FS objectives for the OCA are discussed below.

2.6.2.1 Field Survey

Previous sampling data indicate that lead-affected sediment distributions in the OCA vary according to depositional environment. The two primary depositional environments affecting the distribution of sediments are the active stream channels and overbank areas. In addition, ponding associated with historic beaver dams or culvert structures and high water or flooding events of the Scioto River could have implications for lead distribution.

A field survey was conducted to refine the current understanding of depositional features within the OCA. The survey included mapping the distribution of overbank deposits and low- and high-flow boundaries, and documenting any significant modifications to the fluvial system (e.g., by human or animal activity) that affect depositional processes or the distribution of potential lead-bearing particulates. A second key objective of the mapping was to select actual sediment transect, overbank, and surface water sampling stations for nature and extent evaluations. These features were mapped on the topographic base map prepared for the area at a 1-ft contour interval. During the supplemental remedial investigation sampling activities, additional topographic surveying was conducted to confirm the Site features and locations that were based on the topographic base map originally prepared for the project; no significant topographic changes were documented in the updated survey (see Section 3.2 for additional discussion of survey activity).

2.6.2.2 Sediment and Overbank Sampling

Sediment and overbank sampling was conducted to confirm and complete previous investigation work and historical sampling by Dames & Moore (1991) presented in the work plan (PTI 1995).

Historical sampling locations are presented in Figure 2-8. Remedial investigation sampling stations are presented on Figure 2-9.

During the initial remedial investigation, two cross-channel transects were established in the farm ditch at locations upstream and downstream of the confluence with the north branch of the offsite creek. The farm ditch originates in residential and light commercial areas of southwest Circleville. Two transects were established in the lower reach of the offsite creek in the area of the beaver dams mapped by Dames & Moore (1991). These two transect locations were near areas where former beaver dams may have resulted in sediment deposition. Following the initial field survey and discussions with Richards, two additional transects (T-OCA-5 and –6) were located in an abandoned channel segment south of the current Farm Ditch in the vicinity of the Earnhardt Hill water treatment facility (Figure 2-9). According to Mr. Richards, the southern channel was abandoned in 1979 when he developed the current deeper channel to the north to improve drainage of his farm field. Twelve overbank stations were established during the initial remedial investigation based on results of the field survey conducted to characterize depositional processes and areas.

Samples were collected in 6-in. intervals to a depth of 30 in. All samples were analyzed for lead, and selected samples were analyzed for additional PCoIs. In addition, samples from representative locations were analyzed for grain size, TOC, pH, CEC, lead mineralogy, and percent glass fraction.

Following completion of the initially planned remedial investigation sampling, additional transect and core sampling was conducted during the supplemental sampling in 2003 to further delineate the extent of lead across the OCA extending downstream to the flood control berm on the Farm Ditch near the confluence with the Scioto River. In general, soil/sediment samples were collected along each transect from the centerline and sides of the stream channel, top of bank, and at overbank locations up to a total depth of 48-in. and analyzed for lead in 6-in. depth intervals. Some of the collected soil/sediment samples were archived at the laboratory and were analyzed for lead if the overlying or nearby sample indicated a significantly elevated lead concentration. The supplemental sampling locations are shown on Figure 2-9.

The supplemental TPH investigations were performed in 2005 to determine the presence and extent of TPH within the OCA downstream of the South Ditch. Consistent with the samples collected from the South Ditch, sediment samples were collected from the 0- to 6-in., 6- to 12-in., and 12- to 24-in. depth increments at two locations within the OCA located immediately west of the South Ditch and analyzed for TPH. The samples from the two deeper intervals were held at the laboratory and were analyzed for TPH if the overlying samples contained TPH at a concentration greater than the maximum allowable residual TPH concentration of 5,000 ppm for Type I soils, as defined in OEPA's Division of Emergency and Remedial Response's January 14, 2004, guidance document titled *Soil Leaching to Groundwater Evaluation for Total Petroleum Hydrocarbons* (OEPA 2004a). The TPH sampling locations are shown on Figure 2-9.

Finally, supplemental sediment sampling was performed in support of sediment bioassays and revisions to the ecological risk assessment. Consistent with the samples collected within the South Ditch, sediment samples were collected from the 0- to 6-in. depth increment using an

Ekman dredge. Those samples were collected from certain previous remedial investigation sampling locations in an attempt to collect samples representative of the following concentration ranges: 40 to 400 ppm, 400 to 800 ppm, and greater than 800 ppm. The samples were analyzed for lead, TOC, AVS/SEM, and grain-size. Samples representative of the specified target concentration ranges were selected for subsequent sediment bioassay testing. The sediment sampling locations associated with the sediment bioassay testing are shown on Figure 2-9.

2.6.2.3 Surface Water Sampling

Sampling was conducted to evaluate the potential presence of lead in surface water downstream of the South Ditch, in the offsite creek and farm ditch, and from other point-source discharges to the OCA from surrounding residential and commercial areas. Surface water sample locations included the offsite creek below U.S. Highway 23; the farm ditch below the confluence of the south fork of the offsite creek with the farm ditch; the farm ditch at its confluence with the Scioto River; the farm ditch flowing south from Circleville; and the outfall draining the residential and commercial areas east-southeast of the OCA (Figure 2-10). Seasonal changes and field conditions dictated surface water sampling at individual locations. Sampling at low-flow conditions was conducted in the main channel at stations 1, 2, 3, and 5. Samples were collected during high-flow (storm event) conditions from below culverts discharging to the OCA at stations 4 and 5 (Figure 2-10).

Sampling was conducted first at the most downstream location, then progressed upstream to avoid sediment disturbance that could affect water quality. Samples were analyzed for total and dissolved lead. Measurements of stream flow (discharge), TSS, hardness, pH, electrical conductivity, temperature, and dissolved oxygen were also collected.

2.7 Lead Particle Analyses

This section describes the methods used to evaluate lead mineralogy and the fraction of lead entrained within glass particles in selected soil/sediment samples, including lead concentration as a function of particle size, optical evaluation, and analysis by electron microprobe and computer controlled scanning electron microscopy (CCSEM). These analyses were conducted to further characterize the form and nature of lead and lead-bearing particulates in samples collected during the remedial investigation; these analyses are identified in Table 2-2.

2.7.1 Optical Inspection

For the optical inspection, 5 samples from the East Swale, 4 samples from the South Ditch, and 14 samples from the OCA (Table 2-2) were prepared by oven drying to constant weight at 50°C. Bulk material from each sample was evaluated according to SOP-63b in the work plan (PTI 1995). A Fisher® Stereo Master binocular microscope at 50× magnification was used to scan each sample for 15–20 min., and the quantity, shape, and size of glass particles were recorded.

2.7.2 Electron Microprobe Analysis

Subsequent to the optical inspection, six samples (T-ES-2 B 0-6, T-ES-4 B 0-6, G-SD-1, T-OCA-3 B 0-6, C-OCA-2 6-12, and C-OCA-10 6-12, Table 2-2) that contained either elevated bulk lead concentrations or abundant glass particles in the optical inspection were selected for electron microprobe analysis. Electron microprobe analyses were conducted on the less-than-600-μm size fraction (lower end of coarse sand). This particle size cutoff was necessary to obtain a sufficient number of particles on the grain mount for analysis. Sample preparation procedures and instrument operating parameters are found in SOP-63a of the work plan (PTI 1995). A JEOL 8600® electron microprobe housed at the Department of Geological Sciences, University of Colorado at Boulder, was used to evaluate the six samples for the presence of lead mineral forms and to determine the chemistry of glass particles in these samples. Particle size (i.e., long axis dimension) was recorded for 144 randomly encountered glass particles in the six samples. Quantitative analyses were conducted on 44 of these glass particles using a wavelength dispersive detector to determine the chemical composition of the glass particles. Photomicrographs of representative glass particles were also collected.

2.7.3 Computer Controlled Scanning Electron Microscopy

CCSEM was used to determine the amount of lead entrained within glass particles in 5 samples from the East Swale, 4 samples from the South Ditch, and 16 samples from the OCA (Table 2-2). The CCSEM work was conducted by RJ Lee Group, Inc., Monroeville, Pennsylvania. CCSEM is particularly well suited for identifying and quantifying easily identifiable end-member mineral phases, such as lead-bearing glass particles. Because an SOP for the CCSEM procedure was not presented in the work plan, a description of the analytical approach is provided in Appendix D. The report from the RJ Lee Group is provided in Appendix E.

2.7.4 Lead Concentration as a Function of Particle Size

As part of the OCA investigation, seven of the soil/sediment samples were screened through a 500-μm sieve, and the less-than- and greater-than-500-μm-size fractions (i.e., material that passed through the sieve and material that did not pass through the sieve, respectively) were analyzed for lead concentration. These analyses were conducted to evaluate whether lead in the OCA soils/sediments was distributed primarily within the fine or coarse soil/sediment fractions.

2.8 Data Quality Assurance Review

A quality assurance review was conducted to verify that the laboratory's quality assurance and quality control (QA/QC) procedures were documented and sufficiently rigorous to meet EPA method requirements, laboratory standard operating procedures (SOPs) (where applicable), and the data quality objectives (DQOs) specified for the project. Data validation procedures were based on criteria established in EPA's functional guidelines for evaluating organic and inorganic analyses (U.S. EPA 1994a,b). Data qualifiers were assigned, as necessary, during the quality assurance reviews in accordance with U.S. EPA (1994a,b) and with DQOs established for this

project (see quality assurance project plan [QAPP], Table 3-1, in the work plan [PTI 1995]), and the QAPP for the supplemental remedial investigation sampling event (BBL 2002). Data quality was assessed with respect to DQOs established for this project, including DQOs for completeness, analytical methods (comparability), accuracy, precision, and representativeness. A summary of the analytical methods used to generate data for this investigation is included in Table 3-1 in Appendix B of the work plan and in the supplemental remedial investigation sampling QAPP (BBL 2002).

The following laboratory deliverables were reviewed during the data validation process:

- A case narrative discussing the analytical problems (if any) and procedures
- Documentation of instrument calibrations and frequency of calibration; 28 percent of supplemental remedial investigation sampling data were subjected to this review
- All laboratory-reported compound identifications (e.g., chromatograms and confirmation analyses) and quantification of results; 28 percent of supplemental remedial investigation sampling data were subjected to this review
- Method blanks associated with each sample delivery group (SDG); 28 percent of supplemental remedial investigation sampling data were subjected to this review
- Results of all laboratory quality control check samples, including surrogate compounds, laboratory control samples, matrix spikes, matrix spike duplicates, and laboratory duplicate analyses; 28 percent of supplemental remedial investigation sampling data were subjected to this review
- Sample preparation logs and dilution volumes
- All laboratory-assigned flags and qualifier codes
- Instrument and sample detection limits for all target analyses
- Sample handling procedures and holding times, including chain-of-custody and sample processing and analysis records (e.g., laboratory bench sheets).

In addition, results for field quality control samples were evaluated to provide additional information in support of the quality assurance review. The field quality control results are discussed in the attached quality assurance review summaries (Appendix C). Additional information related to the analyses is discussed below.

An EPA Level I data validation review was conducted for the remedial investigation project data and 28 percent of the supplemental sampling data. The data validation procedures summarized below were performed for all analyses:

- Review of chain-of-custody documentation to verify completeness of the sample set for each data package submitted
- Verification of holding time requirements
- Review of the reported laboratory quality control information to verify the initial and continuing calibration information and results for laboratory quality control samples, including laboratory blanks, laboratory duplicate samples, matrix spike/matrix spike duplicate samples, laboratory control samples (LCSs), surrogate spikes, and other method-specific quality control measures (e.g., analytical spikes, gas chromatography/mass spectrometry [GC/MS] tuning requirements, recoveries, etc.) to determine whether results are within target control limits and DQOs for this project
- Evaluation of the results for field quality control samples
- Assessment of the impact of laboratory and field quality control results and assignment of data qualifiers.

2.8.1 Metals Analyses

Inorganic analyses for solid and aqueous matrices were conducted by inductively coupled plasma-atomic emission spectrometry (ICP-AES) and graphite furnace atomic absorption spectrometry (GFAA), as specified in EPA SW-846 Methods 6010A, 6010B (lead only for supplemental sampling), 7041, 7060A, 7191, and 7421 (U.S. EPA 1991b, 1996b). The sample digestion procedures specified in EPA Method 3050A (U.S. EPA 1991b) were used for sample preparation for soil, sediment, and water samples for SW-846 analyses.

Analyses for TCLP lead were completed on selected samples collected during the supplemental sampling event using EPA Method 1311 and 6010B (U.S. EPA 1992a, 1996b).

2.8.2 Organic Analyses

EPA SW-846 methods (U.S. EPA 1991b) were used to analyze water, sediment, and soil samples for VOCs and SVOCs. TPH analyses were conducted as specified in EPA Method 418.1 (U.S. EPA 1983). Analysis of PCBs was performed as specified in EPA Method 8080 (U.S. EPA 1991a).

2.8.3 Fluoride Analyses

Fluoride analyses were conducted by potentiometric selective ion electrode for aqueous samples as specified in EPA Method 340.2 (U.S. EPA 1983). For solid matrix samples, the fluoride analyses were conducted by EPA Method 340.2 after Bellack digestion.

2.8.4 Lead Mineralogy and Glass Fraction Analyses

Lead mineralogy was analyzed using an electron microprobe equipped with an energy-dispersive detector, as described in SOP 63a (PTI 1995). Bulk glass fraction analyses were determined optically with a binocular microscope at 50× magnification. Appendix D includes a description of the additional CCSEM analyses to further quantify the amount and phase of lead present in selected soil/sediment samples.

2.8.5 Data Quality Assessment

Data summary tables for the data collected during the remedial investigation are included in Appendix B. Data qualifiers assigned to individual results are included in the table. Individual sample results were qualified as undetected (*U*) or estimated (*J*) because of quality control requirements specified in the methods or the DQOs of the project. None of the data generated for the remedial investigation were rejected.

The quality of historical data was assessed during the generation of the project work plan. The historical data were categorized by matrix and study, and the data assessment was documented (PTI 1995).

The quality assurance review summaries for the remedial investigation and supplemental sampling data are included in Appendix C. The corresponding data summary tables are included in Appendix B. The review summaries and data tables include the justification of qualification of specific sample results and a listing of all remedial investigation data.

2.9 Deviations from the Work Plans

Procedures specified by the work plan were followed during the Site investigation. Notable deviations from the 1995 work plan are summarized in Table 2-3. Some of these deviations were additional work scope items described in the July 22, 1997 RI/FS work plan modification request (GE 1997) approved by OEPA. Other minor modifications or variances from the investigation procedures that were determined to have no effect on the data were documented during the field activities, but are not included in Table 2-3. Table 2-3 also includes key deviations for the supplemental sampling completed at the South Ditch and the OCA in 2003, as well as deviations from the scope of sampling specified in the sediment bioassay work plan.

3 Physical Characteristics of the Site

This section describes the physical characteristics of the Site, including climate and meteorology, Site topography, and key geologic and hydrogeologic conditions relevant to the RI/FS.

3.1 Climate and Meteorology

General climatic data for the area are provided in a U.S. Department of Agriculture (USDA) report (USDA 1980) prepared in cooperation with the Ohio Department of Natural Resources (ODNR). The average annual precipitation for the area is 38 in., with the wettest period typically between the months of April and July. Average seasonal snowfall is 13 in. The temperature varies widely, but averages 33°F in winter and 73°F in summer. Prevailing wind in the area is from the south-southwest. The average wind speed is generally highest, approximately 11 miles per hour, in early spring.

3.2 Topography

Aerial photographic surveying and topographic mapping of the Site was completed in 1993 to support the RI/FS. Aerial photography was conducted by Henderson Aerial Surveys, Inc., Columbus, Ohio. Ground-truth topographic mapping was conducted by Patridge & Associates, Inc. (Patridge), also of Columbus, Ohio. The mapping was based on the Ohio State Plane Coordinate System using a vertical U.S. Geological Survey (USGS) datum benchmark established by the City of Circleville. A copy of the Site topographic base map showing 1-ft contours and Site features is provided in the work plan; Figure 1-4 shows 5-ft contours for this area.

Following completion of the remedial investigation sampling in 1996, a local access frontage road was constructed by the City of Circleville across the east end of the offsite creek along the west side of Highway 23. Subsequently, after completion of the interim action activities in 2002 (Section 1.4.8), a second access road was constructed by Richards along the west side of the railroad tracks. Based principally on these developments, additional aerial photographic surveying and topographic mapping were conducted in 2003 to develop a more current base map to support the supplemental remedial investigation sampling activities. The 2003 aerial photography was conducted by M.A.N. Mapping Services, Inc., of Hilliard, Ohio, and Patridge and BBL conducted the associated ground-truth mapping using the same coordinate system and benchmarks as the 1993 survey. This remedial investigation report contains figures utilizing base maps from both surveys; however, the remedial investigation and supplemental remedial investigation soil/sediment sampling stations and results for the South Ditch and OCA areas are all presented on the updated base maps.

The plant is located approximately 700 ft above sea level on broad, gently sloping terraces of the Scioto River Valley. The terraced uplands of the valley contain small streams and unnamed

ditches that drain to the Scioto River, which is located approximately 0.75 miles west of the Site and is approximately 30–40 ft lower in elevation than the terraced uplands that include the plant. One of these ditches (referred to as the South Ditch) flows from east to west on the south side of the plant. The South Ditch originates from drainage of an expansive marshy area beginning approximately 300 ft east of the Site where it receives runoff from a railway, fields, and residential areas; the ditch discharges from the Site to the OCA, which also includes a small marshy area near the confluence with a farm ditch that discharges to the Scioto River. The lower marsh and lower portion of the ditch also receive surface water runoff from U.S. Route 23, a railroad line along the west side of the highway, and adjacent residential, agricultural, and light commercial areas.

3.3 Geology and Soils

3.3.1 Regional Geology

The regional geology of the area consists of interbedded glacial outwash and till deposits that overlie consolidated shale bedrock (Walker et al. 1965). Recent alluvium covers the glacial deposits along stream valleys. The glacial materials were deposited during the Pleistocene Epoch (Illinoian Stage) and consist of clay, silt, sand, gravel, and boulder lithologies that show considerable variation in thickness and composition vertically and horizontally. These deposits include glacial till (heterogeneous mixtures of boulders, sand, silt, and clay) interbedded with outwash deposits (typically sand and gravel layers with some interbedded silt and clay layers) that accumulated in the area during repeated advances and retreats of ice (Norris 1988). The bulk of the rock fragments in the outwash sands and gravels consist of limestone and dolomite pebbles derived from predominantly limestone source rocks in the upper reaches of the Scioto River Basin (ODNR 1965). The glacial processes carved out long valleys in the bedrock and deposited the thick sequence of glacial materials. Norris (1988) found that these glacial deposits ranged in thickness from about 135 to 185 ft in the vicinity of the Site.

Cross section A-A' (Figures 3-1 and 3-2) presents a regional geologic cross section through the Site. The cross section shows the relationship of the glacial deposits to the Scioto River and the inferred depth to the underlying shale bedrock in the vicinity of the Site. Section A-A' was prepared from drilling data at the plant site and well log data from the EHWD wellfield located adjacent to the Scioto River, approximately 0.75 miles southwest of the Site. The glacial deposits range from approximately 100 ft thick near the Scioto River, where they are overlain by recent age alluvium, to 180 ft thick northeast of the plant (Figure 3-2). The underlying shale consists of units of the Olentangy and Ohio Formations of Devonian age (E&E 1989).

A lower sand and gravel unit (outwash deposits) in the glacial deposits represents the principal groundwater aquifer in the Circleville area. The underlying shale bedrock is not utilized for water supply. The highly productive sand and gravel deposits are interpreted to have formed in former glacial meltwater channels and thus may show considerable variation in thickness; one of these former meltwater channels is believed to trend roughly northeast-southwest along the southeast corner of the plant site (Layne 1983). Production wells PW-1 and PW-2 are screened in these sand and gravel deposits. This sand and gravel unit appears to thin considerably toward

the north-northwest and appears to be more highly stratified with clay (or till) in the vicinity of the EHWD wellfield (Figure 3-2).

The lower sand and gravel unit is overlain by a sequence of interbedded sand, sand and gravel, silt, and clay units. Individual units in this sequence are not expected to be regionally continuous. However, the low-permeability clay and silt units are expected to act locally as aquitards that isolate shallow groundwater from deeper productive zones of the regional aquifer.

3.3.2 Site Geology

The current understanding of Site geology is shown in two geologic cross sections referred to as B-B' and C-C'. Cross-section locations are shown in Figure 3-1 and the sections are illustrated in Figures 3-3 and 3-4. These cross sections were developed from existing borehole and well log data compiled from previous geotechnical, water supply, and environmental investigations conducted at the Site, with additional remedial investigation data collected from four boreholes completed around the perimeter of the EFA (Figure 3-5).

The Site is underlain by a 20- to 30-ft-thick sand and gravel unit that comprises the shallow water table zone. This unit may be thickest toward the eastern half of the plant site (MW-1 in Figure 3-3) and appears to include a thin clay interbed (approximately 1–5 ft thick) beneath and immediately west of the main plant area at approximately 20 ft bgs (Figure 3-3). Toward the east and in the vicinity of the EFA (Figure 3-4), the upper portion of this unit appears finer-grained, consisting predominantly of silt and clay deposits. At the EFA, the stratigraphy consists of sandy or silty clay in the upper 4 to 8 ft in boreholes EFA-1, BH-14, BH-15, and BH-16, with sands and gravel with thin interbeds of silt or sandy silt to depth. Boreholes BH-16 and BH-EFA-1 encountered notable clayey silt deposits at depths of approximately 24 ft and 52 ft bgs, respectively.

Throughout the Site, the upper sand and gravel unit is underlain by a thick clay unit that locally separates the shallow aquifer from deeper productive units in the glacial deposit sequence. The clay unit is 20 ft thick in the vicinity of wells PW-1 and PW-2 and is approximately 70 ft thick north of the EFA at borehole TH1-83, where the clay also contains some gravel (possibly till deposits) in the upper portion of the unit (Figure 3-4). The clay unit was intercepted in MW-16 at 51.5 ft bgs, or approximately 650 ft in absolute elevation, along the north edge of the EFA. A clay layer was also intercepted in borehole BH-EFA-1, located at a lower elevation immediately east of the EFA at approximately 24 ft bgs (approximately 663 ft in absolute elevation). This clay was similar to the other clays observed and this material was sampled for analysis of physical properties (Appendix A). Physical property data classify the clay unit as a silty clay with sand with a permeability of 1.7×10^{-7} cm/sec. The characteristics of the underlying clay unit immediately beneath the plant have not been identified and this was not an objective of the remedial investigation; however, the unit appears to extend beneath and to the west of the plant where the clay may contain interbeds of sand or sand and gravel (MW-9 in Figure 3-2).

The thick clay unit is underlain by sand and gravel deposits interbedded with silts and clays. The more permeable sand and gravel deposits in this sequence appear thickest to the southeast,

in the vicinity of the Site production wells, as described above under *Regional Geology*. These sands and gravels form the principal aquifer for water supply in the area. This lower aquifer is approximately 65 ft thick at Site production wells PW-1 and PW-2 (Figures 3-3 and 3-4), and appears to thin significantly toward the north and northwest as indicated in borehole TH1-83 (Figure 3-4) and borehole TH2-83 (Figure 3-3), and it appears to contain more clay interbeds to the west-southwest in the vicinity of the EHWD wellfield (EHWD data projected in Figure 3-2). These productive sands and gravels appear to be underlain by a substantial lower clay unit that overlies bedrock.

3.3.3 Soils

Soils at the Site are loams, sandy loams, and silty clay loams that have developed from glacial outwash and alluvium (USDA 1980). The soils in the farm field south of the South Ditch have been mapped as the Westland silty clay loam with inclusions of the Warsaw loam and Princeton sandy loam soil series. The field to the east of the Thomson production wells has a higher content of organic matter and is mapped as the Carlisle muck soil series (USDA 1980). The soils immediately adjacent to the Site and to the eastern property line have been mapped as "Urban Land"; however, the undisturbed soils would likely be mapped as the Westland silty clay loam because of similar topographic position to the adjacent farm field. The Westland series is a deep, very poorly drained soil with surface horizons of very dark gray, friable, silty clay loam approximately 16 in. thick. The subsoil is typically an alkaline clay loam. The soils in the OCA are predominantly the Shoals silt loam derived from alluvium, which has eroded from the adjacent upland terraces. This is a somewhat poorly drained soil with grayish brown silt loam surface horizons to approximately 34 in., with a brown-to-gray mottled, alkaline subsoil (USDA 1980).

To accommodate OEPA's administrative requirement to determine soil reference area concentrations, soil concentrations of EPA Contract Laboratory Program (CLP) target analyte list (TAL) metals, fluoride, and TPH were determined in soils collected from seven reference area locations. Samples were collected at depths of 3.5–5 ft from shallow backhoe excavations in an area located southeast of the Site (Figure 3-6). This area was unaffected by former facility operations, but may have been influenced by anthropogenic activity, such as historical agricultural activities. Results of the analyses are summarized in Table 3-1 and complete analytical data are provided in Appendix B. Additionally, as discussed with OEPA (OEPA 2009a), the arsenic data for the Adjacent Fields soils was used to represent Site-specific background arsenic concentrations.

3.4 Hydrogeology

3.4.1 Regional Hydrogeology

The Scioto River Valley contains permeable sands and gravels that lie within an ancient stream-cut bedrock valley. The aquifer covers over 320 mi^2 and is up to 8 miles wide and up to 250 ft thick (Walker et al. 1965). Circleville is located at the north end of the valley where the aquifer

averages over 100 ft thick and well yields can exceed 1,000 gal per minute (gpm) (Walker et al. 1965).

In the Circleville area, the regional aquifer is composed of glacial deposits that are 1.5–2 miles wide and extend southwest of the city for approximately 6 miles (Norris 1988). These deposits, described above under *Regional Geology*, range in thickness from 135 to 185 ft in the vicinity of the Site and overlie shale bedrock. As described, these deposits formed from multiple glacial advances and retreats, which left a sequence of sand, sand and gravel, silt, clay, and glacial till deposits in a complex three-dimensional framework. The glacial till or other clay and silt deposits act as semicontinuous aquitards that locally retard the vertical movement of groundwater. Water that infiltrates to deeper parts of the aquifer likely follows selective pathways between and around the local deposits of till, clay, and silt.

The natural movement of regional groundwater in the Scioto Valley is from the valley sides toward the river. Recharge is primarily from precipitation in the valley and recharge from bedrock at the edges of the valley. Only during flooding of the river does water flow back into the aquifer, with these events generally being short lived (ODNR 1965; Norris 1988). Regional groundwater flows from east to west in the vicinity of the Site.

Water supply wells in the aquifer are screened at the base of the outwash deposits where the most permeable sands and gravels are located. The yields to these wells are supplied by a combination of underflow, infiltration, and in some areas induced infiltration from the river (Norris 1988). Induced infiltration is highly variable and dependent on pumping rates and the distance between the wellfield and the river. The EHWD and Du Pont wellfields, which are located adjacent to the river south-southwest of the Site, receive the majority of their water from underflow and infiltration, with only approximately 10 to 20 percent of the water produced from the river (Norris 1988). The Thomson wells, which are located 6,000 ft from the river, do not receive recharge from the river. Pumping from these wells creates a drawdown cone of depression at their location; at the EHWD and Du Pont wellfields, the drawdown includes a component of flow from the Scioto River toward the wells.

Except in the vicinity of the pumping wells, it can be assumed that most groundwater in the valley generally moves with the regional gradient toward the river. The only exceptions to this are shallow groundwater zones, such as the one that occurs at the Site, which are locally isolated by underlying aquitards. The shallow groundwater is affected by near-surface features such as ditches, which can act as either recharge or discharge boundaries. In general, the shallow water is expected to migrate westerly, mimicking the slope of the regional topography, until it either intercepts a stream, ditch, or seep and discharges, or locally infiltrates to the deeper producing zones of the regional aquifer.

3.4.2 Site Hydrogeology

The Site hydrogeology is characterized by a shallow water table zone located approximately 15 ft bgs and the deep regional aquifer described above.

3.4.2.1 Deep Aquifer

Available data on the deep aquifer at the Site suggest a variably thick sequence of permeable sands and gravels in a locally confined setting with regional flow to the west. This interpretation is based on review of available stratigraphic information, shallow and deep aquifer potentiometric levels and flow directions, and interpretations presented in existing hydrogeologic reports for the area. The transmissivity of the deep aquifer is reported as 6,700 ft^2/day with a storage coefficient of 6.6×10^{-5} based on a multiple-well aquifer test conducted at the Site (Layne 1983). When operating, the drawdown cone of influence from production wells PW-1 and PW-2 (Figure 2-2) is expected to cover a large area, including the area beneath the plant.

3.4.2.2 Shallow Groundwater System

The shallow monitoring well network at the Site consists of 12 shallow monitoring wells, installed during previous investigations, and wells MW-14, MW-15, and MW-16R, piezometers (PZ1–PZ6), and staff gauges (SG1–SG7), installed during the remedial investigation (Figure 2-2). Eight of the shallow wells (MW-1 through MW-8) were installed under a previous RCRA monitoring program and provide the bulk of data available for the shallow zone. Four wells (MW-10 through MW-13) were installed prior to the remedial investigation as part of a focused investigation of a surface spill of diesel oil that was confined to the north side of Building "C" (CTL 1992).

During the remedial investigation, groundwater elevation monitoring was conducted monthly for all Site wells, staff gauges, and piezometers between December 1995 and April 1997, with an additional event in August 1997. A summary of all the monthly water level data is provided in Table 3-2. Historical data collected prior to the remedial investigation were also reviewed and were found to be consistent with the data collected for the remedial investigation. The data show that the highest groundwater elevations occur during the spring (May) and the lowest water levels during the winter (December). Groundwater in both the shallow zone and the deeper aquifer system fluctuates about 2–3 ft seasonally.

Figure 3-7 presents shallow groundwater elevations and flow directions on May 7, 1996, which are typical for the Site. Under typical conditions, groundwater flows west-northwest toward MW-4, MW-6, MW-7, and MW-8. This flow path is expected to continue to the west where the water discharges to the Scioto River drainage system. The South Ditch also acts as a local discharge zone for the upper portion of the shallow groundwater along the drainage from approximately SG-7 (near the east end of the EFA) to the west property boundary at Highway 23 (see further discussion below on the groundwater/surface water relationship). The apparent northwest flow path for the western portion of the plant area results from slight mounding of the shallow groundwater beneath the central portion of the plant area. This may have occurred as a result of water losses during former plant operations, and basement areas below the water table may complicate flow patterns in this portion of the Site. In general, the water table surface is much flatter in the eastern portion of the Site and steepens to the west. The horizontal hydraulic gradient varies from 1.9×10^{-3} ft/ft for the eastern portion of the Site and increases to 1.6×10^{-2} ft/ft for the northwestern portion of the Site. The vertical hydraulic gradient (measured at MW-9 and piezometer PZ-6) indicates predominantly upward gradient

conditions during all groundwater monitoring events except for the August 1997 event, which occurred during a period of heavy rainfall. The average upward gradient over the 12 monthly monitoring events was calculated at 1.89×10^{-2} ft/ft, with a maximum vertical gradient of 4.07×10^{-2} ft/ft measured during the June 1996 event.

Additional understanding of the variation in flow conditions in the shallow groundwater zone is provided by elevation contour plots for the five quarterly remedial investigation sampling events at the EFA (Figures 3-8 through 3-12). Groundwater contours from the first quarter event (July 1996) show flow west and southwest toward the South Ditch (Figure 3-8). Groundwater mounding effects in the plant area wells (MW-10, MW-11, and MW-12) appear minimized during this event. As the groundwater elevations slightly decrease during the second quarter (October 1996), a groundwater divide forms, with flow directions indicated to the northwest and the southwest. During the third quarter (February 1997), all wells were not included in the monitoring, but the contours still show continued flow to the west. The fourth quarter of groundwater monitoring (April 1997) shows horizontal gradients flattening out slightly to 0.0013 ft/ft. During the August 1997 quarterly sampling event, the groundwater divide is again evident with flow to the northwest and southwest. There is also change in the gradient near MW-14 and PZ-1 (south of the EFA), possibly caused by the heavy rainfall during the period. During all groundwater sampling events, MW-15 remained in a representative downgradient position from the sludge (glass fines) deposits at the EFA.

The shallow zone appears to be separated from the deep aquifer zone by clay layers or clayey glacial till units. This interpretation is substantiated by available stratigraphic information (Figures 3-2, 3-3, and 3-4) and the shallow water table elevation data that show northwesterly and local southwesterly flow away from the area containing wells PW-1 and PW-2 (Figure 3-5). The groundwater flow and water quality for the shallow water table zone beneath the main plant area have been characterized in previous investigations that are described in detail in the work plan (PTI 1995). Three additional wells (MW-14, MW-15, and MW-16/16R) were installed during the remedial investigation on the north, south, and east side of the EFA to provide groundwater quality and flow information in this area. In addition, six piezometers and seven staff gauges were installed during the remedial investigation to characterize the shallow groundwater interaction with the South Ditch.

The horizontal hydraulic gradient to the northwest and away from wells PW-1 and PW-2, coupled with the upward vertical gradients and underlying clay unit identified in boreholes at the Site, indicates that the shallow groundwater zone is locally isolated from the deep aquifer.

3.5 Surface Water Hydrology

The uplands area around the Site is drained by a simple network of shallow ditches. This network historically consisted of two small spur ditches, the East Swale and an unnamed spur ditch, which flowed to a main ditch referred to as the South Ditch (Figure 1-4). The unnamed spur ditch enters the South Ditch near the former potable water treatment plant and carries primarily agricultural runoff from fields immediately to the south. The East Swale is normally dry; however, during significant precipitation events, the East Swale may receive runoff from the east end of the plant and open fields to the east-northeast. The East Swale was blocked in

1990, and since that time, surface water in the East Swale has been collected and routed to the WWTP as part of the Site surface water management program. The South Ditch flows from east to west along the southern portion of the Site before eventually draining to the OCA and the Scioto River drainage system. The South Ditch originates from drainage of marshy open fields near the railroad tracks located approximately 300 ft east of the Site. Potential discharges to the South Ditch upgradient of the Site include runoff from the railway, agricultural fields, and residential areas.

The South Ditch receives surface water runoff from a portion of the facility and direct discharge from four NPDES outfalls, in addition to the water already in the South Ditch from other upgradient sources. The natural base-flow in the South Ditch was measured upstream of former Outfall 001 (Oil Skimmer Pond) at 300 gpm (approximately 0.4 million gpd) in March 1977 (Chester 1977). Discharge from former Outfall 001 contributed an additional 300 gpm. From August 1990 until 2006, effluent from the new WWTP discharged through the relocated Outfall 001 at an average rate of approximately 1 million gpd. Based on this information, the estimated total flow during this period of plant operation was estimated to be approximately 1.4 million gpd downstream of Outfall 001. Water depth in the South Ditch ranges from approximately 0.5 ft to 2 ft, depending on location and season.

In 1998, maintenance of the South Ditch was necessary to ensure proper sampling at NPDES Outfalls 002 and 003. Small quantities of vegetation with some soil attached were dredged from the ditch during this period. In 1999, maintenance had to be performed due to vegetation severely restricting the flow in the ditch. During this maintenance event, all vegetation with some soil attached was removed from the streambed from just west of Outfall A to approximately 60 ft east of NPDES Outfall 002. In both events, the work was performed during periods of low flow. Bales of straw were placed in the ditch downstream of the dredging to allow settling of disturbed soil. All materials dredged were placed on the overbank and the heavy vegetation between the bank and dredged materials was matted down to prevent erosion of any soil back into the ditch. No dredged material was ever characterized or removed from where it was placed.

The South Ditch flows westward from the Site to U.S. Route 23, where it passes through a culvert under the highway and becomes a small creek (referred to as the offsite creek) in a meandering, well-defined channel in the OCA (Figure 1-4). Approximately 400 ft west of the highway, the creek passes through a second culvert beneath the railroad tracks. Approximately 100 ft west of the railroad culvert, the creek divides into two small channels, one flowing more northerly, and the other flowing to the south. Both channels then join a larger ditch (referred to as the Farm Ditch) that drains fields located on the Scioto River floodplain.

The OCA receives discharges from other areas in addition to flow from the South Ditch. Treated effluent from the EHWD water treatment plant is discharged to the OCA through two outfalls immediately to the north of the EHWD plant. A stream that receives storm water drainage from the Logan Elm Village residential area also flows into the OCA. The Farm Ditch, which flows south and then west approximately 2,000 ft before entering the Scioto River, also receives agricultural runoff and storm water runoff from commercial business areas in southwest Circleville. Beaver dams have periodically impeded the flow of the Farm Ditch at some locations historically; however, the owner of the property currently removes impeding

structures to maintain flow within the ditch. In addition, the OCA property owner/manager has recently constructed a berm across the Farm Ditch approximately 300 ft from the confluence with the Scioto River for flood control. This is an earthen structure with approximately 75 ft of culvert installed to allow ditch drainage beneath the berm.

The offsite creek is a local discharge area for shallow groundwater in the area.

3.5.1 Groundwater-Surface Water Interactions

Groundwater and surface water elevation monitoring was performed for a 1-year period at all existing wells, piezometers, and staff gauges on the Site during the period of plant operation. The elevation monitoring data are summarized in Table 3-2. Hydrographs for monitoring wells, piezometers, and staff gauges were also prepared to assist in data evaluation (Figures 3-13 through 3-15).

During the plant operating period, the staff gauge data show that the surface water levels in the South Ditch remained relatively constant throughout the year due to discharges from the former plant and base flow from marshy areas located east of the Site. The maximum observed fluctuation in water levels was 1.51 ft at SG-5, with the lowest level (672.30 ft) measured in September 1996 and the maximum level (673.81 ft) measured during April 1997. The average fluctuation measured for the staff gauges was 0.85 ft. Surface water elevations were at their highest levels during the spring (May and June) and the lowest levels during the winter (December and January).

The piezometer and monitoring well data show a slightly greater fluctuation in elevation throughout the year when compared to the staff gauge levels. The maximum fluctuation observed in the piezometers was 4.05 ft at PZ-3, with the low level at 676.41 (August 1996) and the high level at 680.46 ft (February 1996). The average fluctuation for the six piezometers was 2.47 ft. The piezometers' water level elevations were at their lowest during the winter (December) and highest in the spring (May and June), which correlate with water levels in the South Ditch. The elevations in monitoring wells MW-4, MW-7, MW-8, and MW-9 show the greatest fluctuation during the year, while the other Site wells show limited fluctuations. The greatest fluctuation in elevation is at MW-8, with the highest elevation during the spring (June) at 674.17 ft and the lowest elevation during the winter (December) at 667.18 ft. The average seasonal fluctuation in groundwater level was 3.04 ft for the 15 wells monitored.

The groundwater/surface water interaction along the South Ditch is shown for each groundwater sampling event at the EFA in Figures 3-16 through 3-20. Groundwater elevations in the piezometers were always as high as or higher than the surface water level measured at the corresponding ditch staff gauge, except for one event during the December 1995 monitoring at PZ-1/SG-1/MW-14. In this area, temporary and localized ponding of water may sometimes occur due to restriction of flow in the South Ditch during higher stage flow; localized ponding may occur due to vegetation debris or restriction of flow through culverts beneath service roadways. This temporary ponding may locally reverse flow direction for a limited area and time.

Water elevation data collected from the monitoring wells, piezometers, and staff gauges confirm that shallow groundwater discharges to the South Ditch along the entire segment of the ditch, extending from near the east end of the EFA to Highway 23; this includes the portions of the ditch containing sediments with elevated lead and/or other PCoIs. Because the shallow groundwater is "entering" (not "leaving") the ditch, groundwater monitoring to evaluate any potential effects on water quality from PCoI-containing particulates in the ditch is not necessary or appropriate.

4 Nature and Extent of Contamination

The discussion of the nature and extent of contamination is based on an evaluation of historical data and data collected during the remedial investigation. Results of previous investigations have already been extensively described in the work plan (PTI 1995) and are not discussed in detail here. This discussion summarizes the information that was collected during the remedial investigation to address the data gaps identified in the work plan (and presented, in Section 2, as investigation objectives for each area). Historical data are integrated in the discussions and interpretations where relevant. Tables and figures referred to in this section summarize the data for each area of interest at the Site (EFA, East Swale, Adjacent Fields, former Oil Skimmer Pond, South Ditch, and OCA). Complete tables of relevant historical data and all remedial investigation data are presented in Appendix B. These data are incorporated into human risk assessment evaluations presented in Section 6. Quality assurance reviews of the data generated by the remedial investigation are summarized in Section 2.8 and described in detail in Appendix C.

4.1 East Fenced Area

The remedial investigation objectives for the EFA are described in Section 2. These primarily included delineation of the extent of sludge deposits, characterization of the shallow groundwater flow conditions, and confirmation that the shallow groundwater was not adversely affected by the sludge deposits.

4.1.1 Extent of Sludge

The horizontal extent of the sludge was determined by excavating shallow (approximately 2-ft-deep) backhoe pits at 27 locations on a grid with approximate 100-ft centers. Additional hand-dug test pits were used to further define the extent of sludge/soil to the east and southeast of the EFA. The sludge, which had been pumped from the former lagoons to the EFA, is visibly distinct from native soils; this distinction allowed for the extent of sludge placement to be visually determined in the field. The approximate locations of the investigation pits and the estimated horizontal extent of sludge are shown on Figure 4-1. The area where the glass fines sludge was placed is concentrated within the fenced area with a smaller area that extends a limited distance southeast of the fence.

The general EFA sludge boundary was confirmed through collection of four soil samples from outside the area of visually identified sludge. Sample locations are labeled as CN-1 through CN-4 on Figure 4-1. The soil samples were collected from approximately 2–3 ft bgs and analyzed for lead at all four locations and for antimony, arsenic, barium, chromium, and nickel at two locations (CN-1 and CN-3). Analytical results from samples collected from all four sides of the EFA are presented in Table 4-1. Lead concentrations ranged from 16.2 to 347 mg/kg with the higher concentration at CN-EFA-3, east of the largest area of concentrated sludge, where the boundary is less distinct. PCoI metals were detected at low concentrations at CN-EFA-1 (west) and CN-EFA-3 (east). Antimony concentrations were 2.7 and 3.9 mg/kg,

arsenic concentrations were 18.7 and 18.6 mg/kg, barium concentrations were 96.5 and 114 mg/kg, chromium concentrations were 14.7 and 17.8 mg/kg, and nickel concentrations were 19.0 and 21.6 mg/kg, respectively.

Test pit results indicate that the area of the sludge deposits is slightly larger than previously estimated and portions of the sludge extend slightly outside the current fence line to the south near MW-14 and to the east where a limited area of mixed soil/sludge was confirmed. The sludge boundary is more distinct on the north, west, and south sides, but is less distinct on the east side where the area of mixed soil/sludge was identified.

The ground surface elevation of the EFA area where the main body of sludge is located ranges from 695 to 700 ft amsl (PTI 1995, Plate 2). The maximum groundwater elevation observed in EFA wells (MW-14, MW-15, MW-16) was 681.93 ft (MW-16, May 1996). The available elevation data indicate that the sludge is not in contact with the shallow groundwater.

4.1.2 Additional Sampling at Station 25D

Five surface soil and two subsurface soil samples were collected in the vicinity of the assumed location of the former Dames and Moore station 25D (Figure 4-1) to confirm that sludge-affected soils in the area are limited in extent and insignificant from a risk-based perspective (see Section 6). Analytical results for these samples are presented in Table 4-1. Lead concentrations ranged from 23.0 to 242 mg/kg.

4.1.3 Sludge Permeability

To estimate the permeability of the glass fines to infiltrating rainwater, the sludge was tested *in situ* using a percolation test. The hydraulic conductivity of the sludge was estimated at approximately 7.0 to 9.0×10^{-3} cm/sec using results from two field percolation tests. These results are consistent with a very fine sandy silt or silty sand deposit and should be considered a rough estimate of the permeability of the sludge material. The sludge is relatively permeable; however, it is capped with approximately 1 to 2 ft of soil and vegetation cover, which affects the rate and amount of precipitation that actually comes in contact with, and percolates through, the underlying glass fines.

4.1.4 Groundwater Sampling Results

Five quarterly groundwater sampling events were conducted at the EFA during the remedial investigation. Groundwater samples were analyzed for total and dissolved PCoI metals and for total fluoride. Monitoring well locations are presented on Figure 2-1. Analytical results are presented in Table 4-2.

Antimony was not detected during any of the quarterly monitoring events and nickel was detected in only one well during the fourth-quarter sampling event. Chromium and lead were occasionally detected at low concentrations in total (i.e., unfiltered) samples only during the first three quarters, with maximum concentrations of 9.7 μg/L and 11 μg/L, respectively. These

detections are believed to be the result of excess turbidity in the samples (e.g., up to 40 nephelometric turbidity units [NTUs] at MW-15 during the third quarter when the only lead concentration was reported). Chromium and lead were not detected in dissolved metals samples collected during these events. Arsenic is present in total and dissolved phases ranging from 1.0 *U* to 8.3 μg/L, with the highest concentration observed at MW-14 and MW-15. Fluoride concentrations range from 0.1 *U* to 1.4 mg/L, with average concentrations by well of 0.4 mg/L (MW-14), 0.96 mg/L (MW-15), and 0.32 mg/L (MW-16). Total and dissolved barium concentrations are similar, with the highest concentration in MW-14 at 362 μg/L. For perspective, reported concentrations of PCoI metals and fluoride were well below federal maximum contaminant levels (MCLs) at all wells during all sampling events.

The results for the fourth quarter sampling for lead, chromium, and nickel are questionable. None of these metals were detected in the total water samples; however, detections were reported for the dissolved (i.e., filtered) analyses. No problems in the quality of the analytical data are apparent, and it appears that the total and filtered samples may have been incorrectly labeled in the field or the bottles somehow switched during laboratory analysis. The results reported for the dissolved analyses are believed to actually represent particulates in the total water sample; however, it is not possible to confirm the label or bottle switching based on the available information.

Because of this uncertainty, a fifth quarter of sampling was conducted to replace the fourth quarter results. During the fifth quarter of sampling, dissolved and total lead and nickel were not detected, with dissolved chromium detected at 7.5 μg/L in MW-14.

The remedial investigation data demonstrate that the glass fines sludge at the EFA has minimal impact on the shallow groundwater PCoI concentrations in the vicinity of the EFA.

4.2 Adjacent Fields

Previous investigations have found that concentrations of lead and other metals in soil and biota samples from fields to the north of the plant were not detected at levels of concern. Elevated levels of fluoride, possibly the result of historical stack emissions, were found in vegetation and animal tissue samples. Soil sampling during the remedial investigation was conducted to confirm that lead and fluoride were not elevated at levels of concern in undisturbed soils along the north property boundary next to the adjacent fields.

Surface and subsurface soil samples were collected from four locations along the north property boundary. Sampling locations and results for lead and fluoride are presented in Figure 4-2. Lead concentrations ranged from 52 to 112 mg/kg in the 0-to-6-in. interval and 15.5 to 32 mg/kg in the 6-to-12-in. interval. Fluoride concentrations ranged from 180 to 430 mg/kg in the 0-to-6-in. interval and 48 to 650 mg/kg in the 6-to-12-in. interval. Results for the remaining PCoI metals are presented in Table 4-3, and results for TOC, pH, and CEC are presented in Table 4-4. Lead concentrations in the surface (0–6 in.) soil samples were similar to historical sampling data from fields immediately to the north, while fluoride concentrations in samples collected during the remedial investigation were higher than in samples collected previously from fields to the north. While the concentrations of lead and fluoride in adjacent field soils are

slightly elevated, they are not at a level of concern from a risk-based perspective (see Section 6). Concentrations of other PCoIs were not elevated.

4.3 East Swale

Previous investigations found, in limited sampling, that concentrations of lead and arsenic are elevated in portions of the East Swale, but only in the surface soil/sediment along the drainage channel. The sampling conducted during the remedial investigation provided a more comprehensive characterization of the East Swale, including determining concentrations of PCoIs not previously analyzed (antimony, nickel, and fluoride).

A total of 48 surface and subsurface soil/sediment samples were collected from four transects in the East Swale. Sampling locations and lead concentrations along the central swale are shown on Figure 4-3, while results for other PCoIs are presented in Table 4-5 and results for CEC, pH, and TOC are presented in Table 4-6. Results from historical soil sampling for total metals in the swale are also included in Table 4-5. Similar to historical data collected, PCoI metal concentrations generally correlated well with lead concentrations, with higher PCoI levels associated with elevated lead concentrations. Results for each transect are summarized below:

- T-ES-1: Eighteen samples were collected from this transect at the south end (lowest point) of the swale. Lead concentration was highest in the 0-to-6-in. (A) soil/sediment sample from the center of the swale (17,600 mg/kg), and generally decreased to 419 mg/kg at 24–30 in. bgs; however, lead concentrations varied with depth between these samples. Lead concentrations decreased with depth for samples from both the lower bank (C1) and upper bank (C2) locations. Maximum lead concentrations for lower and upper bank samples were 2,490 mg/kg and 83.1 mg/kg, respectively. The results indicate that lead has concentrated in the bottom of the swale, and that soil/sediment has possibly been reworked or has experienced a complex history of deposition.

- T-ES-2: Eighteen samples were collected from this transect. The maximum lead concentration (4,640 mg/kg) was detected in the 0-to-6-in. (A) soil/sediment sample collected from the bottom of the swale. Lead concentrations at this location decrease with depth, except for the 12-to-18-in. interval, where lead was slightly higher than at the 6-to-12-in. depth. Samples collected from the lower (C1) and upper (C2) banks of the swale show a decrease in lead concentration with depth from maximums of 169 mg/kg and 191 mg/kg, respectively. Sample results indicate that the lead is concentrated in the bottom of the swale, and that the swale soil/sediment has probably not been reworked as extensively as at T-ES-1.

- T-ES-3: Eighteen samples were collected from this transect. The maximum lead concentration (5,850 mg/kg) was found in the 0-to-6-in. (A) soil/sediment sample collected from the bottom of the swale. Lead concentrations between 6 and 36 in. bgs (B identifiers) are an order of

magnitude (or more) lower than the surface sediment (A identifier) concentration, but do not show a consistent decreasing pattern with depth. These data indicate possible reworking of the soil/sediment or complex history of deposition along the channel of the swale. Lead concentrations in samples collected from the lower (C1) bank of the swale varied with depth, with maximum concentrations on the left (30.1 mg/kg) and right (304 mg/kg) banks both occurring at the 6-to-12-in. interval. Lead concentrations were higher in the upper bank (C2) samples, with the maximum lead concentration of 937 mg/kg in the 0-to-6-in. interval on the plantward side of the swale. The results indicate that while lead has concentrated in the bottom of the swale, some reworking or complex history of deposition of soil/sediment has probably occurred along portions of the swale bank.

- T-ES-4: Eighteen samples were collected from this transect. The maximum lead concentration (5,010 mg/kg) was found in the 0-to-6-in. (B) interval immediately below the soil/sediment in the center of the swale. The lead concentration in the overlying (A) soil/sediment was 1,720 mg/kg. Lead concentrations decrease with depth, but were detected up to 540 mg/kg at 24–30 in., similar to the concentrations observed at this depth in T-ES-3. Samples collected from the lower right bank (C1R) of the swale decreased with depth from a high of 85.1 *J* mg/kg at the 0-to-6-in. interval to 35.6 *J* mg/kg at the 12-to-18-in. interval; however, the lead concentration in the lower left bank (C1L) was slightly higher at the 12-to-18-in. interval (145 mg/kg). Maximum lead concentrations on the upper banks occurred in the 0-to-6-in. intervals (198 mg/kg at C2L and 74.5 mg/kg at C2R), and decreased with depth. As with the other transects, these results indicate that some reworking of soil/sediment along the swale channel probably occurred during operations at the plant.

Remedial investigation data demonstrate that the southern end of the swale contains the highest concentrations of PCoIs; the concentrations observed are generally similar to the limited historical sampling by Dames & Moore. The distribution of the elevated lead along the swale is also generally restricted to the soil/sediment along the bottom of the swale.

The vertical profile of the PCoI metals in the swale is somewhat different from that observed for other areas, suggesting a more complex history of filling or reworking of soils and sediments along and within the swale. The consistent pattern of decreasing lead concentration with depth observed at other areas of the Site is less pronounced or sometimes reversed at locations in the swale.

The bottom of the swale at the south end is approximately 9–10 ft above the highest observed groundwater levels. To the north near TES-4, the bottom of the swale is approximately 13–14 ft above highest groundwater levels. The remedial investigation data indicate that soils/sediments containing elevated lead or other PCoIs within the swale are located approximately 6–11 ft or more above the highest elevation of groundwater observed in the vicinity of the swale.

4.4 Former Oil Skimmer Pond

Previous investigations found elevated levels of lead and petroleum hydrocarbons (hydraulic oil) in the surficial sediments of the pond. These surficial sediments were removed when the pond was excavated and backfilled. Confirmation samples were not collected at the time of excavation. In addition, an oily horizon extending east from the pond was noted in the excavation and in a subsequent test pit. Sampling was conducted during the remedial investigation to fill these data gaps.

4.4.1 Borehole Confirmation Sampling Results

To confirm that previous removal activities at the pond area had excavated deep enough to remove any oil-contaminated soil, two boreholes were drilled to a depth of 12 ft through the mid-portion of the former pond location to obtain confirmation samples of soil beneath the pond backfill materials (Figure 4-4). One soil sample from the bottom of each borehole was submitted for laboratory analysis. Results for the borehole samples are summarized in Table 4-7. Only low levels of two chemicals (TPH and di-*n*-butyl phthalate), possibly associated with the former oil skimming operation, were detected in the samples, as well as low levels of three VOCs (acetone, carbon disulfide, and methylene chloride), which are common sampling and/or laboratory contaminants. Metals were not detected at elevated levels. Remedial investigation data demonstrate that adequate closure measures were previously completed for the pond.

4.4.2 Evaluation of Oily Soil Horizon

Five backhoe test pits were dug to approximately 5 ft bgs to evaluate the possible presence of a thin, dark, oily soil horizon to the east of the former pond location (Figure 4-4). Soil samples collected from each pit were analyzed for TPH; the most highly affected soil (sample CN-OSP-5) was analyzed for VOCs and SVOCs. Results of these analyses are summarized in Table 4-8.

TPH results in soil within and around the layer range from undetected at a detection limit of 12 mg/kg to a maximum concentration of 1,950 mg/kg. In the soil sample with the highest TPH concentration (sample CN-OSP-5 [1,950 mg/kg]), VOCs and SVOCs were not detected except for toluene and xylenes, which were found at low concentrations (4.4 μg/kg and 3.6 μg/kg, respectively).

In addition to the test pit sampling, a Geoprobe® investigation was conducted to the east and south of the former Oil Skimmer Pond location to supplement the test pit work and confirm minimal impacts to the soils in the vicinity of the former pond (Figure 4-4). Soil samples were collected from six borings, and groundwater samples were collected from locations "F" and "G." Soil and water samples were analyzed for VOCs using a field gas chromatography unit. VOCs were not detected in any of the soil or water samples. In addition to the field analyses, two of the soil samples were analyzed for TPH at an offsite laboratory. TPH was detected in samples CN-OSP-6 and CN-OSP-7 at 44 and 43 mg/kg, respectively.

The thin (approx. 0.5–1 ft thick at 4–6 ft bgs) sandy horizon containing hydraulic oil east of the former pond is estimated to measure approximately 50 ft^2. The sampling of the oily horizon indicates that the affected soils are limited in area. Analytical results indicate that the TPH present is from hydraulic oil that does not contain hazardous constituents. This material is expected to naturally degrade with time. No adverse impacts on groundwater are expected.

4.5 South Ditch

Extensive sediment and soil sampling has been conducted previously along the South Ditch (PTI 1995). Lead concentrations in the ditch are generally elevated from the area below the former storm sewer outlet from the East Swale (at approximately 24+00) and extending to the western boundary of the Site; the highest concentrations are observed in the vicinity of current or former outfalls to the ditch and in localized sediment accumulation areas downstream from the outfall. During the most recent sampling, performed in December 2005, TPH was not observed at elevated concentrations (BBL 2006). As described in Section 2, sampling was conducted during the remedial investigation to address specific data gaps in the historical sampling.

4.5.1 Confirmation Sediment/Soil Sampling

To supplement data from previous investigations on the vertical extent of PCoIs in South Ditch sediments, subsurface sediment and soil samples were collected at stations TSD0+25 and TSD19+30 (Figure 2-6). Results of this sampling are presented in Table 4-9. Sampling results from previous investigations are presented in Table 4-10. Results for analysis of sediments for conventional parameters conducted during the remedial investigation are presented in Table 4-2.

Samples were analyzed for the PCoI metals at TSD0+25 in the 0-to-6-in. and 6-to-12-in. intervals, and for lead only at the 12-to-18-in., 18-to-24-in., and 24-to-30-in. intervals. Similar to historical sampling results, concentrations of all PCoIs decreased significantly with depth. These results indicate that the sediments containing lead particles have been deposited in slack water and not reworked.

Samples were analyzed for the PCoI metals at TSD19+30 in the 6-to-12-in. and 12-to-18-in. intervals, and for lead only in the 0-to-6-in., 18-to-24-in., and 24-to-30-in. intervals. At this outfall location, PCoI concentrations initially increased with depth, with the maximum concentrations found at the 12-to-18-in. depth. The concentration of lead decreased significantly below 18 in.; however, the pattern of detection is irregular, and likely reflects disturbances (i.e., reworking) or complex depositional history due to its proximity to the current storm water outfall.

As discussed extensively in the work plan, elevated lead concentrations in the South Ditch are found primarily in the shallow soils/sediments (typically up to 12-in. depth) only. The extent of particulates containing lead and possibly other PCoI metals in the ditch is well characterized.

4.5.1.1 Supplemental Transect Sediment/Soil Sampling

To further refine the vertical and horizontal extent of lead concentrations in soil/sediment of the South Ditch, a total of 88 surface and subsurface sediment and soil samples were collected from six transects (T-SD-1 through T-SD-6) between the former East Swale outlet and the western boundary of the Site (Figure 2-6) in 2003. Along each transect, soil/sediment samples were collected from the approximate centerline of the South Ditch, as well as from the sides of the ditch and the top of the bank. The transect locations were selected to provide a more uniform distribution of lead data along the South Ditch.

Similar to previous sampling events, samples were analyzed for lead in 6-in. depth intervals. The supplemental sampling results are presented in Table 4-11. Integrated remedial investigation, historical, and supplemental sampling results for lead are presented by depth (at 6-in. intervals) on Figures 4-5A through 4-5F. Some subsurface samples were originally archived at the laboratory, and were analyzed for lead if the overlying or a nearby sample demonstrated a significantly elevated lead concentration.

The supplemental remedial investigation data confirm the results of the historical and initial remedial investigation sampling. The highest lead concentrations were generally detected in the soil/sediment samples collected near current or former outfalls to the South Ditch, in localized sediment accumulation areas downstream from the outfalls, and at the western end of the ditch where particles have settled above the culvert beneath the highway. The elevated lead concentrations are primarily found in the shallow soils/sediments (typically occurring at depths of less than 1 ft).

Testing for TCLP lead was conducted on two samples from transect T-SD-2. As summarized in Table 4-11, lead concentrations in these samples did not exceed the RCRA regulatory level of 5 mg/L for lead as identified in 40 CFR 261.24.

4.5.2 Sediment Sampling for Organic Compounds

As described in Section 2, sediment samples were initially collected and analyzed for organic compounds from two locations (G-SD-2 and G-SD-3). These samples were analyzed for TPH, VOCs, SVOCs, and PCBs to support analyses being conducted at the former Oil Skimmer Pond and to confirm that the media containing hydraulic oil do not contain hazardous constituents at significant concentrations along the South Ditch downstream of the former Oil Skimmer Pond outfall (Figure 1-2). TPH and nine PAHs as well as low levels of 2-butanone and carbon disulfide, were detected in both sediment samples (G-SD-2 and G-SD-3) (Table 4-12). PCBs were not detected in either of the samples and therefore were excluded from further analyses as discussed in the work plan.

Because PAHs were not detected in oily materials known to be associated with the former Oil Skimmer Pond, the TPH materials in the South Ditch containing PAHs are believed to be from a different source. Historical sampling (Dames & Moore 1989) indicated TPH values up to 7,600 mg/kg at approximately 19+30 ft on the South Ditch near the storm water outfall from the east end of the plant (storm sewer outlet B, Figure 1-2). To determine whether there were multiple sources for the historical TPH detections in South Ditch, additional samples were

collected from both banks of the ditch at station 19+30 and 0+25 (Figure 2-6). A total of 12 PAHs were detected at stations 0+25 and 19+30, and carbazole was also found at station 19+30 (Table 4-13). Based on available historical data and additional sampling conducted, the source of TPH containing PAHs is believed to be the storm water outfall near 19+30.

At the request of OEPA, supplemental TPH investigation activities were performed in the South Ditch in 2005. The 2005 investigation included the collection of sediment samples from eight locations within the portion of the South Ditch adjacent to the former plant site (Figure 2-6).

The results of the supplemental TPH investigation were transmitted to OEPA in the February 17, 2006, *Supplemental Investigation Report* (BBL 2006). As indicated therein, the observed TPH concentrations (ranging from undetected to 250 ppm) in the surficial sediment samples (0- to 6-in.) were significantly lower than historic TPH concentrations observed within the South Ditch (ranging from undetected to 250,000 ppm). Based on those results, and in accordance with the *Scope of Work for Supplemental Investigation Activities* (BBL 2005), the samples from the underlying depth increments (6- to 12-in. and 12- to 24-in.) were not subject to analysis. The data from this supplemental investigation are presented in Table 4-14. The supplemental investigation report was accepted by OEPA in a letter to GE dated March 24, 2006.

Results for sampling of conventional parameters (CEC, pH, TOC, grain size) in South Ditch sediment are presented in Table 4-15.

4.5.3 Upstream Sediment and Surface Water Sampling Results

Upstream (background) sediment and surface water samples were collected from the South Ditch near the east property boundary (Figure 2-6; Station G-SD-1). Surface water samples were collected during both high-water and low-water events. Results are summarized in Tables 4-16 and 4-17.

PCoI concentrations in background sediment samples were not elevated with the possible exception of fluoride, which was detected in the field duplicate sample at 3,000 *J* mg/kg, as compared to 210 *J* mg/kg in the primary sample (Table 4-16). Total and dissolved lead were not detected in the background surface water sample from either the low-flow event in December 1995 or the high-flow event in May 1996 (Table 4-17).

Supplemental upstream (background) sediment samples were also collected during the supplemental TPH and sediment bioassay investigations. Specifically, sediment samples were collected from two locations during the supplemental TPH investigation: one location adjacent to the eastern property boundary; and a second location further upstream (i.e., east), as shown on Figure 2-6 (Stations SD-I and SD-J).

The results of that supplemental TPH investigation were included in the February 17, 2006, *Supplemental Investigation Report* (BBL 2006). As indicated therein, both surficial (0- to 6-in.) sample results were undetected, as indicated in Table 4-16. Based on those results, and in accordance with the *Scope of Work for Supplemental Investigation Activities* (BBL 2005), the samples from the underlying depth increments (6- to 12-in. and 12- to 24-in.) were not analyzed.

Regarding the supplemental sediment bioassay investigation conducted in 2007, four upstream (background) surficial sediment samples were collected: two from locations upstream of the eastern property boundary; and two from the drainage ditch next to the adjacent farm fields, which drains to the South Ditch south of the EFA (samples T-BG-4-A, T-BG-5-OB1R, T-BG-6-A, and T-BG-7-OB1L). These samples were analyzed for lead, TOC, AVS/SEM, and grain size. As indicated in Table 4-18, the lead concentrations were consistent with background, with concentrations ranging from 14.1 ppm to 82.1 ppm. Certain of these samples were selected for subsequent sediment bioassay testing, which is further discussed in Appendix G.

4.6 Offsite Creek Area

Previous investigations at the OCA have found elevated levels of lead in sediments and soils, presumably the result of transport from the South Ditch. The lead-bearing particulates had been observed to be concentrated in limited areas of the OCA. Field activities conducted during the remedial investigation were designed to refine this delineation of the horizontal and vertical extent of the lead-containing particles, and to confirm the stability or lack of further migration of the lead-bearing particulates within the OCA.

4.6.1 Field Survey

A field survey was conducted to refine the understanding of depositional features within the OCA. This information was used to guide remedial investigation sampling locations and to assist in interpretation of all OCA data collected. The survey included the mapping of the areas where overbank deposits were likely to occur, areas expected to be within high-flow and low-flow boundaries, and any significant modifications to the offsite drainage system that may have affected depositional processes or the distribution of potential lead-bearing particulates. The results of the mapping are shown on Figure 4-6. The different depositional environments mapped include the primary stream channels, overbank, and marsh/cattail areas. The majority of the OCA area was mapped as an overbank depositional environment with narrow channels. Small areas described as marsh exist near the transition area from the South Ditch to the relatively flat area of the OCA.

4.6.2 Soil/Sediment Sampling

Soil and sediment samples were collected from six transects and twelve core locations during the remedial investigation to supplement data from previous investigations. Historical and remedial investigation sampling stations are presented in Figures 2-8 and 2-9, respectively. Remedial investigation transect and core sampling results are presented in Tables 4-19 and 4-20, respectively, and historical streambed and overbank sampling results are presented in Tables 4-21 and 4-22, respectively. Results for conventional parameters from remedial investigation sampling are presented in Table 4-23. Integrated remedial investigation and historical results for lead are presented by depth (at 6-in. intervals) on Figures 4-7A through 4-7E.

Field reconnaissance surveying and evaluations of lead concentrations confirm work plan assumptions that the high lead concentrations are generally restricted to the offsite creek and adjacent overbank areas between Highway 23 and the railroad tracks, and the broader depositional area between the railroad and the Farm Ditch where sediment particulates have settled. The remedial investigation data also confirm that high lead levels in the broader depositional setting are generally restricted to the overbank areas or small channels reworking these overbank deposits; lead concentrations are significantly lower in the major active channels and all other locations at the OCA below the affected areas mentioned.

In general, the high lead levels at the OCA are also restricted to the upper 6 in. to 1 ft of the soil/sediment profile. The maximum lead levels (5,000 to 10,000 mg/kg) were concentrated in the 0-to-6-in. interval in a roughly triangular-shaped area where the offsite creek splits immediately west of the railroad tracks (Figure 4-7A). Lead concentrations immediately around this area of maximum concentration were in the range of 1,000 to 5,000 mg/kg. A secondary (higher water) channel paralleling the Farm Ditch to the south contains some elevated lead (500 to 1,000 mg/kg range) in surficial sediments only (additional transects T-OCA-5 and T-OCA-6 were located in this channel). Based on information provided by the landowner (Richards) during the initial field survey, this southern channel was the primary drainage ditch before it was abandoned in 1979. In 1979, the owner dredged what is now recognized as the current ditch (to the north of the original channel) to a deeper level (lower elevation) to improve drainage of the farm field on the north side of the ditch. The abandoned channel is dry most of the year; during higher water or flood events, water within the channel drains back into the Farm Ditch immediately below the T-OCA-6 location.

The areas of elevated lead in the 6-to-12-in. interval are shown on Figure 4-7B. These areas are significantly smaller and contained within the areas where lead is elevated at the 0-to-6-in. depth (Figure 4-7A). The elevated lead concentrations for the 12-to-18-in. interval are further restricted in area and also contained within this same area immediately west of the railroad tracks (Figure 4-7C).

Field observations indicate that the owner/manager of the OCA property actively maintains the Farm Ditch to ensure continuous drainage. As noted above, a significant portion of the current ditch was formed by excavating at the boundary of the farm field in 1979. Sampling within and along the ditch transects has shown that lead concentrations are generally similar where sampled upstream and downstream from the region of higher concentrations in the depositional area below the railroad tracks (i.e., deltaic area). These data demonstrate that the routine maintenance activities by the farmer have not significantly affected the transport of lead-bearing particulates downstream.

No beaver dams or other natural obstructions were observed to result in significant settling of sediment particles between the high lead area near the railroad tracks and the Scioto River. The OCA property owner/manager has recently constructed a berm across the Farm Ditch approximately 300 ft from the confluence with the Scioto River for flood control. This is an earthen structure with approximately 75 ft of culvert installed to allow ditch drainage beneath the berm. Despite the presence of this berm, the OCA was observed to flood and become thoroughly inundated by higher water on more than one occasion during the remedial investigation sampling.

4.6.2.1 Supplemental Transect Soil/Sediment Sampling

To further refine the vertical and horizontal extent of lead concentrations in soil/sediment of the OCA, a total of 1,519 surface and subsurface soil/sediment samples were collected at 47 transects (T-OCA-7 through T-OCA-53) and 11 core locations (C-OCA-13 through C-OCA-23) during supplemental sampling in 2003 (Figure 2-9). The transects are approximately 100 ft apart and were selected to provide a more uniform characterization of lead concentrations across the OCA. Along each transect, soil/sediment samples were collected from the approximate centerline of the drainage channels, as well as from the sides of the channel, top of the bank, and across adjacent overbank areas. Due to field observations, a limited number of additional samples were collected at high water mark locations (T-OCA-30 through T-OCA-46) to provide more complete delineation along this segment where the channel bank was higher than in other areas. Ten of the cores (C-OCA-13 through C-OCA-22) were established within the deltaic area and one core (C-OCA-23) was located north of the deltaic area (Figure 2-9).

Similar to previous sampling events, samples were analyzed for lead in 6-in. depth intervals. Supplemental sampling results are presented in Table 4-24. Integrated remedial investigation, historical, and supplemental results for lead are presented by depth on Figures 4-7A through 4-7E (in 6-in. intervals). Some subsurface samples were originally archived at the laboratory, and were analyzed for lead if the overlying or a nearby sample demonstrated a significantly elevated lead concentration.

The supplemental data confirm the results of the historical and earlier remedial investigation sampling, including the fact that elevated lead concentrations in the OCA are found primarily in the depositional areas (deltaic area and western portion of the upper creek). The highest lead concentrations were found in the deltaic area, with the maximum lead concentration (15,800 mg/kg) detected in the 6- to 12-in. interval of T-OCA-23-OB1R. Elevated lead concentrations are also present in the western portion of the upper creek located between Highway 23 and the railroad tracks. Less elevated lead concentrations occur in limited locations above and below these areas, with downstream locations generally limited to overbank soils/sediments, and one older abandoned channel, to the south and east of the current Farm Ditch, which is typically dry. In general, the highest lead concentrations were observed in surface sediments and decreased with depth, with most occurring at depths of less than 1 ft.

TCLP testing for lead was run on supplemental samples from T-OCA-15-C1R and T-OCA-27-OB1R. Full TCLP analysis was performed on samples from C-OCA-13-A/B and C-OCA-21-A/B. Table 4-24 summarizes the results for TCLP lead. None of the TCLP results exceeded RCRA regulatory levels as identified in 40 CFR 261.24.

4.6.2.2 Supplemental TPH Investigation

As previously indicated, supplemental TPH investigation activities were performed at the Site in December 2005. The investigation included the collection of sediment samples from two locations within the portion of the OCA adjacent to the railroad tracks, as shown on Figure 2-9.

The results of the supplemental TPH investigation were included in the February 17, 2006, *Supplemental Investigation Report* (BBL 2006). As indicated therein, the surficial (0- to 6-in.) sample results ranged between 25 and 52 ppm, as shown in Table 4-14. Based on those results,

and in accordance with the *Scope of Work for Supplemental Investigation Activities* (BBL 2005), the samples from the underlying depth increments (6- to 12-in. and 12- to 24-in.) were not analyzed.

4.6.2.3 Supplemental Sediment Bioassay Sampling

In support of the ecological risk assessment presented in Section 7 and Appendix G, supplemental sediment sampling was performed within the OCA in May and June 2007 pursuant to the OEPA-approved sediment bioassay work plan (Exponent 2007). The bioassay sampling involved the collection of surficial sediment samples (0- to 6-in.) from 28 locations previously sampled during the remedial investigation in an attempt to collect samples representative of the following concentration ranges for use in the sediment bioassays: 40 to 400 ppm, 400 to 800 ppm, and greater than 800 ppm. The sediment sampling locations associated with the sediment bioassay testing are shown on Figure 2-9. Samples were analyzed for lead, TOC, AVS/SEM, and grain size. The data for these samples are presented in Table 4-18. Certain samples representative of the specified target concentration ranges were selected for subsequent sediment bioassay testing, which is further discussed in Appendix G.

4.6.3 Surface Water Sampling

Surface water samples were collected in the OCA from four locations during low-flow conditions and from two locations during high-flow conditions (Figure 2-10). During the low-flow sampling event (December 1995), station SW-OCA-4 was dry. During the high-flow sampling event (May 1996), samples could not be collected from stations SW-OCA-1, -2, and -3 due to unsafe conditions. Results from these sampling events are summarized in Table 4-25.

Dissolved lead was not detected in any of the surface water samples. Low concentrations of lead were detected in the total water samples only at SW-OCA-5 (6.3 μg/L low flow to 13.5 μg/L high flow) below the South Ditch, and in the low-flow discharge at SW-OCA-1 (at 3.4 μg/L) where the Farm Ditch discharges to the Scioto River. These data indicate some particulate transport of lead at very low concentrations is occurring from the South Ditch and, to a lesser extent, through the OCA.

Until 2006, the flow in the offsite creek was maintained year-round by the continuous discharge to the South Ditch from the Thomson plant. Therefore, estimates of discharge from the South Ditch to the OCA did not vary significantly from the assumed low-flow event of 2.6 ft^3/sec (cfs) (December 1995) to high-flow event of 3.1 cfs (May 1996). In addition, for the high-flow event, field sampling station SW-OCA-5 was moved approximately 50 ft downstream from the low-flow location (i.e., out of the highway culvert), which could affect comparison of flow estimates and data quality.

The lead concentrations detected during this time were lower than allowable concentrations for Thomson's NPDES permit for discharge to the South Ditch. The lead concentrations detected are also well below Ohio water quality standards for protection of aquatic organisms. Both the

dissolved and total lead concentrations are below the corresponding water quality criteria[1] (30 and 37 μg/L, respectively) for the protection of aquatic life at a hardness of 400 mg/L $CaCO_3$.

4.6.4 Upstream Sediment Sampling

In support of the ecological risk assessment presented in Section 7 and Appendix G, supplemental sediment sampling was performed in the upstream portion of the farm ditch north of the OCA as part of sediment bioassay investigations. Those investigations involved the collection of surficial sediment samples (0- to 6-in.) from the three upstream (background) locations shown on Figure 2-9 (Stations T-BG-1-A, T-BG-2-OB1R, and T-BG-3A). Samples were analyzed for lead, TOC, AVS/SEM, and grain size. As shown in Table 4-18, the observed lead concentrations (ranging between 26.2 and 77.6 ppm) were generally consistent with background concentrations. Certain of these samples were selected for subsequent sediment bioassay testing, which is further discussed in Appendix G.

4.7 Lead Particle Analysis Results

This section presents results from the lead mineralogy and the glass fraction analyses described in Section 2.7. Optical inspection, electron microscopy, and analysis of lead concentrations as a function of particle size were conducted to characterize the nature and form of lead and lead-bearing particulates in soils/sediments at the Site.

4.7.1 Optical Inspection

Results from the optical inspection of bulk soil material indicate that the quantity of glass particles in the 28 samples evaluated ranged from relatively abundant to not observed (Table 2-2). Glass particles were generally angular to subangular regardless of sample location, although some rounding of fragments was typical of offsite glass particles compared to onsite ditch and swale samples. Glass particles could not be further quantified optically due to the presence of weathered quartz grains, which are nearly identical to the glass particles when viewed under a binocular microscope.

4.7.2 Electron Microscopy

Results from the quantitative analysis of the less-than-600-μm glass particles by electron microprobe indicate that high-lead (funnel glass, up to 24 percent lead) and low-lead (panel glass, up to 3 percent lead) particles contain 15.4±3.2 and 2.2±0.5 weight percent lead, respectively, in the six samples evaluated (Table 4-26). The remainder of the glass is composed primarily of silicon with minor amounts of potassium, calcium, aluminum, manganese, and iron. Glass particles in these samples were quite small; almost all were less than 10 μm in size. Photomicrographs of representative high- and low-lead glass particles (Figure 4-8) demonstrate

[1] These criteria are the values for outside the mixing zone, obtained from the Water Quality Criteria for the Ohio River Drainage Basin, 3745-1-34, effective October 31, 1997.

the angular to subangular glass particle shapes, indicating that glass particles have been resistant to weathering since emplacement in Site soils/sediments.

In addition to lead in glass particles, common soil lead forms were observed in all six samples. These include lead sulfide, iron-lead oxide, manganese-lead oxide, lead phosphate, and barium-lead sulfate, indicating that some portion of the lead in Site soils exists as mineral precipitates, rather than being entrained in a vitrified matrix. It should be noted that all of the electron microprobe and CCSEM results pertain to the less-than-600-μm size fraction. (The distribution of lead in the coarse fraction of the samples is further discussed in Section 4.7.3.)

As with the electron microprobe analyses, glass particles observed by CCSEM were invariably quite small, with the majority of glass particles less than 3 μm in diameter (Figure 4-9, Table 4-27). Occasionally, leaded glass particles up to 100 μm in diameter were observed in selected samples. No obvious differences in glass particle sizes were observed in samples from the East Swale, South Ditch, and OCA. For the glass fraction, lead was present mostly in the high-lead category, rather than the low-lead category for all samples analyzed (Table 4-27). Samples with higher bulk lead values often contained greater concentrations of lead in glass than in other mineral forms (Table 4-27). However, on average, lead in glass accounted for 5, 10, and 17 percent of bulk lead concentration in the fine fraction (i.e., less-than-600-μm) samples from the East Swale, South Ditch, and OCA, respectively.

In summary, the lead mineralogy analysis indicates that 5–17 percent of lead in the less-than-600-μm size fraction of samples from the East Swale, South Ditch, and OCA is entrained in glass particles. This lead is generally present within small (less than 3 μm in diameter), high-lead (i.e., 15.4±3.2 weight percent) glass particles. The remaining lead in the fine fraction of these samples is composed of common soil alteration phases (sulfides, sulfates, oxides, and phosphates) and lead adsorbed on the surfaces of fine sediment particles (i.e., iron hydroxides, organic matter, silts, and clays).

4.7.3 Lead Concentration as a Function of Particle Size

Results from the analysis of lead concentrations in the less-than- and greater-than-500-μm size fractions in the seven OCA soil/sediment samples indicate that lead concentrations are nearly identical for these two size fractions in each sample (Table 4-26). These results are inconsistent with the general behavior of lead in soils, wherein lead is typically concentrated in the fine soil fraction (Duggan and Inskip 1985; Zimdahl and Skogerboe 1977). Normalizing the concentration data for the mass of soil in each size fraction indicates that the majority of lead mass in these OCA samples is present in the coarse fraction (88 percent on average, Table 4-28). Because lead is typically concentrated in the fine soil/sediment fraction, due to the presence of small mineral precipitates and lead adsorbed on soil surfaces, these data indicate that a majority of the lead mass in the coarse soil/sediment fraction in the OCA (88 percent on average) is present as a nonstandard lead form. It is possible that this lead is present entrained in glass.

5 Transport and Fate of PCoIs

Physical transport (i.e., sediment transport) is the dominant mechanism by which PCoIs are transported at the Site. The primary physical transport mechanism is entrainment of PCoI-bearing sediments by surface waters, and movement of these particulates along the active drainages (principally the South Ditch) to the OCA (Figure 1-4). Other potential transport mechanisms include chemical transport mechanisms that involve the dissolution of PCoIs in groundwater or surface water and transport in the dissolved phase; however, these transport mechanisms appear to be insignificant at the Site. The potential transport mechanisms, the data that indicate which mechanisms are active at the Site, and a conceptual model of PCoI fate and transport are discussed in the following sections. The analyses presented in this section are based on remedial investigation data collected prior to the supplemental sampling for lead completed in 2003. The supplemental data generally confirm the previous data, and further assessment of the supplemental data in this context would not significantly change the interpretations originally presented in the draft April 1998 remedial investigation report.

5.1 Physical Transport Mechanisms

Physical transport is the movement of PCoIs that are bound to particulates that are in suspension in water. Physical transport is the primary transport mechanism for PCoIs at the Site. The dominant physical transport route is through the South Ditch drainage system. The ability of surface water to transport sediments through the ditch system depends on 1) the size and density of the particles being transported, and 2) the velocity and turbulence of the water. Smaller, less dense particles are more easily transported than are larger, denser particles, and an energetic flow system (e.g., a fast, turbulent stream) will better transport particles than will a low-energy flow.

5.1.1 Sediment Particle Size

Seven samples collected in the OCA during the remedial investigation were passed through a 35-mesh (500-μm) sieve, and both the less-than-500- and greater-than-500-μm fractions were analyzed for total lead content by EPA Method 6010A. The 500-μm criterion was selected to meet the data goals of the risk assessment, but it is also useful in characterizing particle size distribution for evaluation of transport mechanisms. The results indicate that 84–94 percent of the sediments sampled in the OCA have a particle diameter of greater than 500 μm (Table 4-28), and that 85–92 percent of the lead mass in the sediments is associated with the greater-than-500-μm fraction. In the sediment samples that contained more than 500 mg/kg lead, the lead concentration in the less-than-500-μm fraction was slightly higher than in the greater-than-500-μm fraction. This relation suggests that the density difference between the lead-bearing and non-lead-bearing minerals may have resulted in a slight bias toward higher lead concentration in the finer particle sizes relative to the larger sizes (smaller particles with higher lead concentrations would have higher densities, and would be more likely to be deposited than smaller, lead-free particles).

The distribution of particle sizes among the clay, silt, sand, and gravel fractions was also analyzed in six sediment samples, three from the South Ditch and three from the OCA (Table 5-1). In general, the sediments collected onsite in the South Ditch are finer than those collected from the OCA, suggesting that once the finer sediments are entrained (i.e., in suspension), they are more likely to remain in suspension (see further discussion below).

5.1.2 Flow Through the Ditch System

The energy of water flowing through the ditch system depends, in part, on the slope (gradient) of the streambed. As the water flows over steeper slopes, its velocity increases, as does its ability to transport sediment. As the slope of the streambed decreases, flow velocity is reduced, and the water can no longer carry as much sediment as it did in the steeper reaches. This relation is evident in the sediment-lead concentration data collected during the remedial investigation and previous investigations. By comparing the maximum lead concentrations observed in sediments at each station along the South Ditch and OCA with the stream gradient (Figure 5-1), it is apparent that lead-bearing sediments have been deposited in areas where the streambed's gradient lessens (e.g., upstream of the highway and railroad culverts). In particular, a triangular area of higher lead-bearing sediments has been deposited west of the railroad tracks, where the slope is very shallow (Figure 4-5). This area is referred to as the OCA deltaic area. Within a short distance downstream from the deltaic area (less than 400 ft), observed lead concentrations in soil/sediment decrease rapidly, indicating that under most streamflow conditions, the shallow slope of both the deltaic area and the farm ditch mitigates further transport of the lead-bearing sediments away from the deltaic area.

A second factor that can affect the energy and velocity of waters as they flow through the ditch system is the smoothness of the creek bed bottom. In those areas where the bed of the channel is rough or highly irregular, eddies and pools may form, within which the velocity of the stream may be sufficiently low to allow the sediments to be temporarily deposited. These types of deposits occur sporadically along the South Ditch and along the upper reach of the offsite creek between the Site and the deltaic area of the OCA.

A third factor than can affect the energy and velocity of the waters flowing through the ditch system is the cross-sectional area of the stream. Increases in the cross-sectional area result in decreases in stream velocity. Consequently, when the banks of a stream are topped during flooding events, the cross-sectional area is greatly increased, and the velocity of the water flowing through these areas is reduced, particularly within the low-lying, flat floodplain. The resulting areas of sedimentation are called overbank deposits.

Although some of the PCoI-bearing sediments were deposited within the South Ditch, the majority of these sediments appear to have been transported to the OCA deltaic area. Because the lead concentrations in the "deltaic" sediments are among the highest concentrations observed, and are also elevated at greater depth than either the streambed sediments or the overbank deposits, the majority of the lead mass in sediments is associated with the "deltaic" deposits.

In summary, PCoI-bearing sediments have been deposited in three regions: 1) temporarily within active channel bars or as bank storage along the South Ditch and in the offsite creek

above the railroad tracks, 2) where the offsite creek slowed due to a decrease in gradient west of the railroad tracks (i.e., the OCA deltaic area), and 3) overbank areas in a limited region of the OCA below the tracks. Routine maintenance of the Farm Ditch appears to have no significant effect on the distribution or transport of limited PCoI-containing sediment downstream.

5.2 Chemical Transport Mechanisms

Chemical transport is the movement of PCoIs that are dissolved in water. Several interdependent variables affect the transport of PCoIs via the chemical transport mechanism:

- The solubility of the PCoI source material (e.g., cullet, glass fines) under the Eh-pH conditions of the water with which it comes in contact
- The composition of the water
- The composition of the soil, sediments, or aquifer materials with which the possible dissolved PCoIs may come in contact.

As further discussed below, the Site data and analyses completed indicate that the chemical transport mechanism is not an important process for PCoI transport at the Site.

The PCoIs defined for the facility, with the exception of petroleum products (i.e., hydraulic oil), occur (or occurred) in both vitrified and nonvitrified forms. The vitrified forms of the PCoIs are essentially inert at the Eh-pH conditions at the Site surface and in the groundwater, so they would not be subject to chemical transport. This conclusion is supported by direct measurements of PCoI solubility from EFA sludge material (i.e., glass particles from the finishing process) using EPtox and TCLP tests, which indicate that only a small fraction of total PCoIs in the vitrified materials are subject to dissolution (Table 5-2). For example, three sludge samples collected by OEPA (1991) contained between 2,900 and 8,800 mg/kg total lead; but only 7 mg/L lead was leached from one of these samples (Table 5-2) and no lead (detection limit of 2 mg/L) was detected in the other two leachates. Similarly, arsenic concentrations released during the EPtox testing of these same samples ranged from 0.07–0.50 mg/L. Because the extraction solution used during the EPtox test, acetic acid, is far more aggressive at liberating metals than is infiltrating rainwater, the EPtox test likely overestimates the solubility of the PCoIs from the sludge, and consequently the mobility of lead and arsenic from the sludge is significantly less than that indicated by the EPtox results. These data are confirmed by the groundwater monitoring results, which do not show elevated metals levels.

Although the PCoIs in the nonvitrified raw materials used at the facility (litharge [PbO], barium carbonate [$BaCO_3$], sodium antimonate, arsenic trioxide [As_2O_3], potassium dichromate [$K_2Cr_2O_7$], bunsenite [NiO], and hydrofluoric acid [HF]) have the potential to dissolve in water, the chemistry of the Site's surface waters, groundwater, stream sediments, and aquifer materials limits their transport. Once dissolved, the PCoIs will either reprecipitate as secondary minerals [e.g., PbS, $Pb_5(PO_4)_3Cl$, $PbCO_3$, $BaSO_4$, $Cr(OH)_3$, CaF_2], or adsorb onto the clays and/or oxy-hydroxides contained within the sediments and aquifer materials. Petroleum products used at the facility, which have been found at the Former Oil Skimmer Pond and in the South Ditch,

have relatively low water solubility and will tend to be sorbed by organic matter present in the ditch sediments.

To assess the capacity of soils/sediments to bind PCoIs via sorption, the CEC, TOC content, and pH of samples from the East Swale, South Ditch, and OCA were determined (Table 5-3). These data indicate that the TOC content of the soils/sediments ranges between 10,000 and 40,000 mg/kg (1–4 percent), the pH ranges between 6.4 and 7.9, and the CEC ranges between 2,000 and 21,000 mg sodium/kg soil. Using the methods of Zimdahl and Skogerboe (1977), the empirical relationships between the soil/sediment pH, CEC, TOC, and the capacity of the soil/sediment to adsorb or fix lead can be calculated (Table 5-3). The average gross lead fixation capacity of the soils/sediments (i.e., the total amount that can be adsorbed based on the method of Zimdahl and Skogerboe (1977) is approximately 18,000 mg lead/kg soil/sediment, and is significantly greater than the average total lead concentration of 3,400 mg/kg observed for data collected through the initial remedial investigation sampling. Consequently, even if the conservative assumption is made that all the lead present in the soils/sediments is present as adsorbed lead and not as vitrified particles, the sediments in the ditch system have an average excess capacity to adsorb approximately 14,000 mg lead/kg soil. These data indicate that if lead were to be released into the surface waters as a dissolved load, the stream sediments would have a large capacity to adsorb that lead. Given the large capacity of the ditch sediments to adsorb lead, it is unlikely that the lead would be transported away from the Site as a dissolved constituent; rather, it would be adsorbed to sediments (or precipitated as secondary mineral phases) and transported via the physical transport mechanisms described in the previous section.

Current Site data demonstrate that dissolved lead concentrations are minimal in both groundwater and surface water (Tables 4-2 and 4-25). In addition, mineralogy data (Section 4.7.2) indicate that secondary minerals (e.g., lead-barium sulfate, lead sulfide) have formed in the sediments of East Swale, South Ditch, and OCA further binding lead in particulate form. Furthermore, a comparison between the lead and arsenic concentrations in both the EFA soils and ditch system sediments (Figure 5-2) indicates that they are strongly correlated (R^2=0.78). Since dissolved lead in most natural systems occurs as a positively charged cation and arsenic occurs as a negatively charged anion, the processes that control their chemical transport as dissolved species are vastly different. If the chemical transport mechanism were responsible for the transport of these PCoIs to their current location in the sediment, the correlation between the total lead and arsenic concentrations would not be expected to be strong. These data, together with the groundwater and surface water concentration data and mineralogy data, support the conclusion that the chemical transport mechanism is not an important process for PCoI transport at the Site.

Given that the OCA is a discharge zone for shallow groundwater and that some areas within the OCA contain elevated concentrations of lead (Figures 4-7A through 4-7E), the groundwater must first flow through these lead-bearing sediments before entering the OCA. If chemical transport mechanisms were important at this Site, then lead would be detected at elevated concentrations in surface waters of the OCA downstream of the lead-bearing sediments. Because they are not, it may be concluded that the lead-bearing sediments are chemically stable. This conclusion is supported by the observation that the lead detected in groundwater and surface water samples during remedial investigation sampling was present only in particulate form (i.e., present in total rather than dissolved samples [Table 4-25]).

5.3 Conceptual Model of Transport and Fate of PCoIs

5.3.1 Transport of PCoIs Via Groundwater

The data presented in the work plan (PTI 1995) demonstrated that groundwater in the vicinity of the plant was not significantly impacted by PCoIs or PCoI-containing soils/sediments. One data gap identified for the remedial investigation was the collection of additional groundwater quality data downgradient of the EFA, where glass fines from the former Lagoons 1 and 2 were placed during the 1970s. To confirm that the EFA sludge has not affected groundwater quality, the total and dissolved PCoI concentrations in wells MW-14, MW-15, and MW-16 were measured during five sampling events between July 1996 and August 1997 (Table 4-2). As discussed in Section 4, data from the fourth sampling event (August 1997) were considered anomalous and were replaced by a fifth event; therefore, the fourth quarter sampling results are not considered further. These wells were installed to monitor any potential effects of the EFA materials on groundwater quality, and were placed immediately adjacent to, and downgradient and/or cross-gradient from, the EFA (Figures 3-8 through 3-12). Dissolved lead was not detected in any of the 16 groundwater samples collected from these wells (detection limit of 0.003 mg/L). Detectable total lead was found in only 1 of 16 measurements, at 11 μg/L in MW-15; however, high turbidity (40 NTUs) was reported in the final water sample and this value is believed to represent particulate detections. The only PCoIs that were consistently detected in the shallow groundwater were barium and fluoride, which averaged 160 and 500 μg/L, respectively (Table 4-2). As mentioned previously for perspective, all detections in groundwater at the EFA are below federal MCLs.

These data indicate that the fate of PCoIs through the groundwater system is either being controlled by the low solubility of PCoIs in the original source materials (i.e., they have not dissolved and migrated to groundwater to any appreciable degree), or that interactions between the aquifer materials and the PCoIs as they potentially dissolve result in their rapid binding as adsorbed species or secondary minerals. Given the geochemical factors discussed above and in the work plan (PTI 1995), and the pattern of PCoI detections in soils (i.e., very limited vertical extent) and groundwater, PCoI-bearing materials from the EFA have not, and are not expected to, adversely affect groundwater quality at the Site.

5.3.2 Transport of PCoIs Via Surface Water

Based on knowledge of historical operations and the pattern of historical and remedial investigation sampling results, the primary pathway for migration of PCoIs at the Site is sediment transport in the South Ditch. Potential historical sources of PCoIs from the Site to the South Ditch are the discharges via storm water drainages and the former outfall from the Oil Skimmer Pond (i.e., hot-end rinse water). As discussed in the work plan, arsenic (42 μg/L), lead (3 μg/L), and fluoride (800 μg/L) were detected in the effluent from the Former Oil Skimmer Pond (Chester 1977), indicating that low concentrations of these PCoIs were discharged to the South Ditch from this source. Also, storm water outfall samples collected by Chester (1977) and Dames & Moore (1993) contained 17-20 μg/L total arsenic, 30–3,300 μg/L total lead, and 300–700 μg/L total fluoride, indicating that low concentrations of these PCoIs were historically

discharged to the South Ditch via surface water runoff. Prior to construction of the wastewater treatment plant and storm water management improvements in 1990, storm water that had the potential to come into contact with process and possibly waste materials stored in uncovered areas at the east end of the property may have conveyed PCoIs to the East Swale. From 1990 until plant closure, these waters were conveyed to the onsite WWTP, eliminating these sources to the South Ditch.

Using all available information and fate-and-transport analyses completed, the following conceptual model of PCoI migration via surface water and sediment transport was developed:

1. During historical storm events, PCoI-bearing materials were occasionally transported to the East Swale and storm sewer system by surface water runoff. The primary transport mechanism was believed to be physical transport of suspended particulates that contained PCoIs; movement of limited amounts of dissolved PCoIs (from the more soluble PCoI-bearing materials) also may have occurred. In addition, low concentrations of petroleum products appear to have been discharged to the South Ditch from the outfall at 19+30.

2. During historical operations, low concentrations of PCoIs were discharged to the South Ditch via the Oil Skimmer Pond outfall, primarily as fine particulates in suspension.

3. Dissolved components, if any, which may have entered the South Ditch, rapidly precipitated out as secondary minerals (e.g., $PbBaSO_4$) or adsorbed onto the sediment substrates.

4. Given that the flow rate from the Former Oil Skimmer Pond was likely much lower than the relatively more energetic baseflow in the South Ditch, most of the suspended sediments discharged were likely transported in suspension downstream without significant deposition in the South Ditch.

5. PCoI-bearing sediments that have accumulated within the East Swale, South Ditch, and OCA (upper creek and deltaic areas) are likely to have been derived primarily from major storm water discharges to the ditch and migration during high-flow events along the ditch.

6. The significant reduction in PCoI-bearing sediments west of the OCA deltaic area indicates that significant transport of PCoI-bearing sediments away from this area is probably not occurring presently. Thus, the current distribution of PCoI-bearing sediments principally occurred during historical discharge events, although minor redistribution of PCoI-bearing particulates may be occurring today during intense storm events.

This conceptual model for the fate and transport of PCoIs via surface water and sediment is consistent with the current distribution of PCoIs at the Site. These observations suggest that significant PCoI-containing sediment is no longer being discharged to the South Ditch from the facility. Limited amounts of PCoI-containing sediment remaining in the South Ditch may

continue to be transported; these sediments appear to be deposited principally in the upper creek and deltaic area of the OCA, probably during major storm water runoff events. Routine maintenance of the Farm Ditch appears to have no significant effect on the distribution or transport of limited PCoI-containing sediment downstream.

6 Human Health Risk Assessment

6.1 Introduction

This section of the remedial investigation report presents the baseline human health risk assessment (HHRA) for the Site. The baseline HHRA evaluates the potential for adverse human health effects from exposures to chemicals of interest (CoIs) under current and potential future Site conditions, in the absence of any action to control or mitigate exposures to these chemicals (i.e., under the no-action alternative). This information is used to estimate the potential health effects associated with Site-related chemicals and to determine whether remedial actions are needed to mitigate any predicted effects.

Figure 6-1 presents the conceptual site model (CSM) developed for this HHRA. A preliminary CSM was provided in the RI/FS work plan (PTI 1995) and was approved by OEPA. The preliminary CSM was then supplemented to reflect additional data collected during the remedial investigation and a revised draft CSM (PTI and BBL 1997) was submitted to and approved by OEPA. The CSM was subsequently updated to include the hypothetical residential exposure scenario for the OCA requested by OEPA following submittal of the April 1998 draft remedial investigation report.

In accordance with EPA guidance, the CSM identifies the primary Site-related chemical sources, environmental release mechanisms, transport routes and media, exposure pathways, and potential human receptors that were considered in preparing the HHRA. Both current and potential future conditions were considered in developing the CSM, which provides the framework for performing the site-specific risk assessment. As discussed in Section 1.2, anticipated future uses of the prior industrialized portion of the property will be consistent with the current zoning regulations (i.e., commercial/industrial) as planned activity and use limitations will preclude possible future residential use of the prior industrialized portion of the property. The risk assessment was conducted in accordance with the general methods described in EPA's *Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual Part A* (U.S. EPA 1989) and in supplemental EPA guidance (U.S. EPA 1991a, 1997, 2003a). The risk assessment approach applied in the HHRA also is consistent with the requirements of the HHRA components of OEPA's generic statement of work for performing an RI/FS, and with OEPA's Division of Emergency and Remedial Response Guidance (http://www.epa.ohio.gov/derr/rules/guidance.aspx).

The HHRA is organized according to the four steps common to most risk assessments:

- Data evaluation
- Exposure assessment
- Toxicity assessment
- Risk characterization.

The principal tasks associated with these four steps are summarized in Figure 6-2. Each of these steps is described in the following sections. Section 6.2 reviews Site data that are relevant to the HHRA and identifies CoIs for human health. Section 6.3 presents the exposure assessment for the Site, including characterization of the exposure setting, identification of exposure pathways, and quantification of exposure for pathways of concern. Section 6.4 presents the toxicity assessment, which includes a discussion of the basis for the toxicity values used to assess risks. Section 6.5 presents the quantification and characterization of human health risks. An analysis of uncertainties associated with the development of risk estimates is presented in Section 6.6.

In preparing the HHRA, efforts were made to focus analyses on the most important contributors to Site-related risks and, thus, those factors that are the likely determinants of remedial decision-making at the Site. As an initial step, PCoIs and potential exposure scenarios were identified. In subsequent analyses, chemicals and exposures that are likely to contribute negligibly, if at all, to Site-related risks were identified, and a list of CoIs and potential exposure pathways was developed for inclusion in the quantitative human health risk analyses. For example, exposure pathways were eliminated from further consideration if an exposure pathway was incomplete. Screening-level analyses were also used to confirm that certain chemicals, exposure pathways, or Site areas are unlikely to contribute significantly to Site-related risks. For example, where initial screening efforts indicated that chemical concentrations measured in a Site area were negligible relative to standard risk-based screening levels, such Site areas were excluded from consideration in quantitative risk analyses. All analyses conducted to evaluate the potential contributions of CoIs and exposure pathways to Site-related risks are documented in this HHRA.

6.2 Data Evaluation

A detailed discussion of the issues that may be considered in identifying CoIs and in organizing data into a form appropriate for human health risk assessment is presented in the data evaluation chapter of EPA's *Risk Assessment Guidance for Superfund* (U.S. EPA 1989). These steps include compiling and evaluating all available Site data for the media of interest; evaluating the analytical methods, quantification limits, and other data quality issues that influence the suitability of the data for use in risk assessment; and developing a list of CoIs and a set of data for inclusion in the risk assessment.

6.2.1 Evaluation of Site Data

Data for various environmental media at the Site are available both from historical Site investigations and from efforts undertaken as part of the remedial investigation, including supplemental data collected in 2003. Relevant historical Site data were included for use in the HHRA if they were determined to be acceptable based on the data usability evaluation presented in the work plan (PTI 1995; Section 1.A.4.1). In addition, all of the remedial investigation and supplemental sampling data providing concentrations of CoIs in appropriate environmental media were used in the HHRA, except for lead data in sediment of the OCA and South Ditch that was collected in 2007 in support of the sediment bioassays, performed as part of the ecological risk assessment. The quality of the data collected during the remedial investigation

was determined on the basis of quality assurance and quality control procedures established as part of the data collection efforts. Remedial investigation procedures for new data collection included consideration of appropriate laboratory analyses and adequate quantification limits.

The available data were used to identify chemical concentrations at specific exposure points of concern (i.e., exposure point concentrations [EPCs]) as required for the analyses undertaken in the exposure assessment portion of the HHRA. The methods and data used to calculate EPCs for specific environmental media of concern are discussed in Section 6.3.4, *Quantifying Exposures*. Copies of the data sets compiled for use in this HHRA and the summary statistics used to develop EPCs, where appropriate, are shown in Appendix F.

As described in the RI/FS work plan (PTI 1995), information from historical plant operations and waste management practices was used to identify the following seven areas of interest at the Site: the EFA, the East Swale, the Former Oil Skimmer Pond, the South Ditch, the OCA, the Adjacent Fields, and the Onsite Soils. Based on the historical data presented in the work plan, the Onsite Soils area was determined to have negligible concentrations of PCoIs and was excluded from the remedial investigation and further risk evaluations in the preliminary CSM. This conclusion is documented below in the screening evaluation.

Data collected during the remedial investigation (including the 2003 supplemental sampling), in combination with historical data, indicate that concentrations of PCoIs present in soil in other areas (e.g., the Adjacent Fields, EFA soils, and Former Oil Skimmer Pond) are also negligible. As a result, these areas were also eliminated from detailed analyses. As previously indicated and discussed with OEPA (OEPA 2009a), the arsenic data for the Adjacent Fields soils was used to represent Site-specific background arsenic concentrations. However, the EFA was retained for evaluation due to elevated concentrations of selected PCoIs measured in sludge materials beneath the soil cover. These evaluations are described in more detail in Section 6.2.2, *Identifying Chemicals and Areas of Interest*.

The initial list of PCoIs for the Site, as presented in the work plan, includes antimony, arsenic, barium, chromium, lead, nickel, fluoride, and TPH. As documented in the work plan, these PCoIs were identified based on an understanding of materials used and wastes generated at the facility, and review and analysis of an extensive database of sampling results from previous investigations at the Site. As part of the remedial investigation, additional data needs were identified and the data were collected as described in the approved work plan (PTI 1995). Supplemental data for lead were also collected in the South Ditch and OCA in 2003. A review of the remedial investigation sampling results confirmed the list of previously identified PCoIs and identified additional PCoIs (i.e., certain carcinogenic PAH [cPAH] compounds) for evaluation in the HHRA, based on remedial investigation sampling at the South Ditch. To refine the list of CoIs for evaluation in the HHRA, all PCoIs for which EPA has developed quantitative toxicity factors or other benchmark values were considered in a human health risk-based screening process. In some cases, where a chemical was not detected in any sample of a specific environmental medium (e.g., antimony and nickel in groundwater), that chemical was excluded from risk-based screening for the specific medium. The procedures used in this screening process and the results of the evaluation are described in the next section.

6.2.2 Identifying Chemicals and Areas of Interest

To screen PCoIs in Site soil, sludge, and sediment, Site data were compared with risk-based concentrations (RBCs) for soil that were developed by EPA (U.S. EPA 2004[2]). These risk-based concentrations are based on exposure to soil via incidental ingestion and are calculated using standard EPA toxicity values and default exposure parameters. As discussed in Section 6.3.1, *Characterizing the Exposure Setting*, an occupational scenario and scenarios involving intermittent and infrequent exposures are most relevant for the Site. As a health-protective element of the screening process, however, RBCs derived for both occupational and residential scenarios were considered in the comparison. The target risk level associated with EPA's RBCs for carcinogenic chemicals is an incremental cancer risk of one in one million (1×10^{-6}) (i.e., a one-in-one-million additional probability that an individual may develop cancer over a 70-year lifetime as a result of the exposure conditions evaluated). The standard EPA target hazard index of 1 is incorporated into the RBC values calculated based on noncancer health effects, indicating an exposure level at which adverse health effects are not expected to occur. A standard RBC is not available for lead. Instead, EPA has identified a screening level of 400 mg/kg for use in assessing whether additional evaluation is required at sites with lead in soil (U.S. EPA 1994d). This value was used in the screening process.

To perform the screening, the RBCs were compared with the maximum observed concentrations of each PCoI in each Site area and environmental medium. Use of the maximum observed concentration as the basis for comparison adds another conservative (i.e., health-protective) element into the screening process. Exposure point concentrations that a receptor might encounter would reflect some combination of the range of observed concentrations and would typically be lower than the maximum value. The RBCs used in the evaluation are shown in Tables 6-1 through 6-9, together with the maximum observed concentrations in each Site area. Additional factors, such as comparisons to background levels or frequency of detection, were also considered in determining the CoIs. As previously indicated and discussed with OEPA (OEPA 2009a), the arsenic data for the Adjacent Fields soils was used to represent Site-specific background arsenic concentrations (i.e., the former fields immediately north of the Site, Table F-8). Using this data set, along with the most current OEPA methodology for calculating appropriate background comparison statistics (OEPA 2009b), an upper cutoff value for arsenic

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[2] The RBCs developed by EPA Region III based on standard EPA default assumptions and algorithms were used in this screening process. After these screening analyses were performed, OEPA expressed a preference for use of the preliminary remediation goals (PRGs) developed by EPA Region IX as the basis for screening analyses in risk assessments performed under their review. Because they are based on a similar set of assumptions and calculations, these PRGs are similar to the Region III RBCs, differing only in the exposure pathways included in the calculations. Use of the Region IX PRGs in place of the Region III RBCs would not change the results of the screening process, nor the risk analyses and conclusions reached in this HHRA. As a result, the screening process presented within the main text of the HHRA has not been modified. At OEPA's request, the results of a screening analysis using the EPA Region IX PRGs (U.S. EPA 2002d) are provided in Appendix H. This appendix also discusses the implications of the results of the alternative screening process for the risk analyses presented in the HHRA. Since the original screening analyses were performed using the EPA Region III RBCs available in 1997, the oral reference doses and RBCs for chromium have been slightly modified, resulting in a slight change in the original results of the screening analyses if the updated values are used. These changes also do not alter the results of the screening process, nor the risk analyses and conclusions reached in this HHRA, and are discussed in Appendix H.

of 33 mg/kg was calculated. The results of the screening process for each Site area and medium are discussed below.

6.2.2.1 Screening Evaluation for Soil, Sediment, and Sludge

East Fenced Area—Twenty-three soil samples were collected from the EFA during the remedial investigation and previous investigations. Samples from this area were analyzed for the inorganic PCoIs. As shown in Table 6-1, with the exception of arsenic, all observed concentrations were well below the comparison risk-based screening levels. Arsenic was observed in 6 of 16 samples at concentrations ranging from 10.0 to 28.9 mg/kg. Thus, the arsenic concentrations for soils observed in this area are below the calculated background upper cutoff value of 33 mg/kg. As a result, risks associated with the arsenic concentrations observed in this area would be comparable to those associated with background concentrations. Based on these results, soil in the EFA was determined to be unlikely to contribute significantly to Site-related risks and was eliminated from further evaluation in the HHRA as an exposure medium of interest.

As discussed in the work plan, the "sludge pits" containing glass polishing and grinding fines were covered with approximately 2 ft of soil in October 1980. Thus, these materials are currently covered and the soil data from the EFA accurately represent the current conditions for exposure potential in this area. As a conservative element of the risk evaluations, however, the sludge data collected from the EFA were also evaluated in this screening process.

Thirteen sludge samples were collected from the EFA during the remedial investigation and previous investigations. Samples from this area were analyzed for the inorganic PCoIs. As shown in Table 6-2, concentrations of barium, fluoride, and nickel were less than the risk-based screening levels. The maximum concentration of antimony exceeded the risk-based screening level based on a residential scenario by approximately a factor of 2. Arsenic and lead also exceeded the risk-based screening levels. Based on these results, the sludge in the EFA was retained for additional evaluation in the HHRA. Antimony, arsenic, and lead were retained as CoIs for this area. As noted previously, the oral reference doses and corresponding EPA Region III RBCs for chromium have been modified since the original screening analyses were performed using the RBCs available in 1997. The values used in the original screening were 390 mg/kg for residential soil and 10,000 mg/kg for an industrial soil for chromium(VI), and 78,000 mg/kg for residential soil and greater than 1,000,000 mg/kg for industrial soil for chromium(III), the more likely form in environmental samples. These values were all greater than the maximum chromium concentration measured in EFA sludge (362 mg/kg). Therefore, chromium was not included in the quantitative risk analyses conducted in the original HHRA.

Following the modifications to the EPA oral RfDs and corresponding RBCs, the RBC for chromium(VI) in residential soil (230 mg/kg) is now slightly less than the maximum chromium concentration measured in EFA sludge. The RBCs for chromium(III) and for industrial soil remain greater than the maximum EFA sludge concentration. Screening against an RBC for residential soil for chromium(VI) represents a highly conservative approach for screening the EFA sludge data because of the unlikely land use scenario and because environmental samples are more likely to contain the less toxic form of chromium, chromium(III). Therefore, because only 2 of the 13 sludge samples had chromium concentrations that exceeded this single criterion

(i.e., chromium(VI) in residential soil), this chemical is excluded from the quantitative risk analyses. As a conservative approach, however, a discussion of the implications of including this compound in the HHRA risk calculations is included in Appendix H.

East Swale—Ninety soil samples were collected during the remedial investigation and previous investigations. Samples from this area were analyzed for the inorganic PCoIs. As shown in Table 6-3, all observed concentrations of barium, chromium, and nickel were well below the comparison risk-based screening levels. The concentrations of fluoride exceeded the risk-based concentration based on a residential scenario at only a single location. All other concentrations of this chemical were well below the risk-based screening levels. Antimony, arsenic, and lead exceeded their respective risk-based screening levels at a greater number of locations. As reported in the work plan, public access to the East Swale is prevented by the security fence around the Site. Anticipated future commercial/industrial uses for the prior industrialized portion of the property will also restrict public access. Furthermore, the East Swale was not part of any Thomson operations and was not visited by Thomson employees on a routine basis. As a conservative element of these risk evaluations, however, the East Swale was retained as an area of interest for additional evaluation in the HHRA based on these results. Antimony, arsenic, and lead were retained as CoIs for this area.

Former Oil Skimmer Pond—Thirteen samples were collected in this area during the remedial investigation. These included two confirmation borehole samples from below the excavation area of the Former Oil Skimmer Pond, five confirmation samples from the oily sandy horizon located to the east of the former pond location, and six Geoprobe® samples from the south and east of the former pond location. The samples from this area were analyzed for the inorganic PCoIs, TPH, VOCs, and SVOCs. As shown in Table 6-4, all concentrations of PCoIs, with the exception of arsenic, were less than the risk-based screening levels. Two samples from this area were analyzed for arsenic. Observed arsenic concentrations were 10.8 mg/kg and 11.8 mg/kg, which are below the calculated background upper cutoff value of 33 mg/kg. As a result, risks associated with the arsenic concentrations observed in this area are comparable to those associated with the local background concentrations.

TPH was also observed in samples from this area. With the exception of a single observed concentration (i.e., 1,950 mg/kg observed in a sample from test pit B), TPH concentrations ranged from 12 to 105 mg/kg, concentrations that are less than minimum regulatory screening levels for TPH in the State of Ohio (Oliver et al. 1996). As reported in the work plan, the source of TPH at the Site is highly refined hydraulic oil, which is characterized as nonhazardous. In addition, during the remedial investigation, chemical analyses were undertaken to determine whether toxic constituents of TPH (e.g., benzene) or other VOCs or SVOCs were present in samples from this area. Acetone, carbon disulfide, di-*n*-butyl-phthalate, methylene chloride, and toluene were analyzed for and observed in at least one Former Oil Skimmer Pond sample; however, the observed concentrations were well below the risk-based screening levels for these chemicals. Thus, based on the nature of the TPH source at the Site and the toxic constituents and other data from this area and the South Ditch, the TPH at the Site was eliminated from further consideration as a CoI. Based on these results, the Former Oil Skimmer Pond was determined to be unlikely to contribute significantly to Site-related risks and was eliminated from further evaluation in the HHRA as an area of interest.

South Ditch—One hundred twenty-one soil samples representing the South Ditch were evaluated in the screening presented in the draft April 1998 remedial investigation report. An additional 64 samples for lead from the supplemental sampling conducted in 2003 are also included in this evaluation. Ten additional samples were collected from the South Ditch in 2005 during the supplemental TPH investigation. Finally, four additional samples were collected from upstream (background) locations associated with the South Ditch in 2007 in support of the sediment bioassay and revision to the ecological risk assessment. The samples collected during these investigations were analyzed for the inorganic PCoIs, TPH, PAH compounds, TOC, AVS/SEM, and/or grain size.

As shown in Table 6-5, observed concentrations of most PCoIs were well below the comparison risk-based screening levels. Concentrations of antimony, arsenic, lead, and certain cPAH compounds exceeded their respective risk-based screening levels in some samples. Some elevated concentrations of TPH were detected in samples collected from the South Ditch during the initial sampling events. However, the levels of TPH observed during the 2005 sampling event were generally lower (ranging from undetected to 250 ppm) than those observed during previous sampling events (concentrations up to 250,000 ppm). Further, as previously discussed, the source of the TPH at the Site was classified as nonhazardous when the plant was operational. In addition, with the exception of cPAH compounds, sample analyses for VOCs and SVOCs found no toxic constituents in this area. Based on this information, TPH was eliminated from further consideration as a CoI. Because concentrations of some PCoIs exceeded comparison risk-based screening levels, the South Ditch was retained as an area of interest for additional evaluation in the HHRA. Antimony, arsenic, lead, and certain cPAHs were retained as CoIs for this area.

Offsite Creek Area—Two hundred seventy-five soil/sediment samples were collected from the OCA during the remedial investigation and previous investigations. Samples from this area were analyzed for the inorganic PCoIs and TPH. Additionally, 965 soil/sediment samples were collected during the supplemental sampling event conducted in 2003 and analyzed for lead. Two samples were collected from the OCA as part of the supplemental TPH investigation conducted in 2005, and 31 samples (including three upstream [background] samples) were collected from the OCA as part of the supplemental bioassay investigation conducted in 2007. The latter samples were analyzed for lead, TOC, AVS/SEM, and grain size. As shown in Table 6-6, with the exception of antimony, arsenic, and lead, all observed concentrations were well below the comparison risk-based screening levels. TPH was observed in one of five samples at a low concentration of 26.8 mg/kg. As discussed in Section 6.3.1, *Characterizing the Exposure Setting*, use of risk-based screening levels based on residential or industrial use provides a conservative basis for comparison. The likely potential exposure scenario for this marshy area is a recreational scenario, involving infrequent potential exposures by older children and adults. Nonetheless, based on these results, the OCA was retained as an area of interest for additional evaluation in the HHRA. Antimony, arsenic, and lead were retained as CoIs for this area. Additionally, at the request of OEPA (OEPA 2002), an evaluation of hypothetical residential exposures to lead was also conducted for the OCA, even though it is within the 100-year flood plain of the Scioto River. The results of the evaluation for a hypothetical residential land use are presented in Appendix I.

Adjacent Fields—Sixty-one soil samples were collected from this area during the remedial investigation and previous investigations. Samples from this area were analyzed for the inorganic PCoIs, with lead and fluoride targeted for additional analysis in the remedial investigation. As shown in Table 6-7, with the exception of arsenic, all observed concentrations were well below the comparison risk-based screening levels. Arsenic was detected in 11 of 16 samples. Concentrations generally ranged from 6.0–32 mg/kg, with a single measured concentration at 113 mg/kg. These arsenic concentrations were similar to those observed in a large multisite study that found background concentrations across Ohio ranging from 1.6 mg/kg to 71.3 mg/kg (Vosnakis and Perry 2009). Thus, the concentrations observed in this area are generally within the range of concentrations observed in Ohio background soil samples. As a result, risks associated with the arsenic concentrations observed in this area would be comparable to those associated with background concentrations. Based on these results, the Adjacent Fields were determined to be unlikely to contribute significantly to Site-related risks and were eliminated from further evaluation in the HHRA as an area of interest. As previously indicated and discussed with OEPA (2009a), the arsenic data for the soils from this area were used to represent Site-specific background concentrations unaffected by facility operations, and were therefore used for screening arsenic data from other areas.

Onsite Soils—As reported in the work plan, the Onsite Soils were determined to have negligible concentrations of PCoIs. As a result, this area was excluded from additional remedial investigation sampling. This conclusion was confirmed in the HHRA screening process. Sixteen soil samples had been collected in this area during previous investigations and analyzed for the inorganic PCoIs and TPH. Antimony, chromium, and TPH were not detected in any sample. Arsenic was detected in one sample at a concentration of 31.3 mg/kg. This concentration exceeds the RBC for industrial soil, but is below the calculated background upper cutoff value of 33 mg/kg. As a result, risks associated with the arsenic concentrations observed in this area would be comparable to those associated with background concentrations. As shown in Table 6-8, all other measured concentrations of PCoIs were below the risk-based screening levels.[3] This evaluation confirms conclusions reached in the work plan that the Onsite Soils have negligible concentrations of PCoIs and this area is not an area of interest.

Summary of Screening Evaluation for Soil, Sediment, and Sludge—In summary, based on comparison of Site data to conservative risk-based screening levels, the following areas of interest and corresponding CoIs were retained for further evaluation in the HHRA:

- EFA sludge: antimony, arsenic, and lead
- East Swale soil/sediment: antimony, arsenic, and lead

[3] In the historical data set for the Onsite Soils, two lead concentrations exceeding the screening level of 400 mg/kg were observed. One of these (5,060 mg/kg) was collected from an area from which soils were subsequently removed as part of clean closure of the former lagoons (PTI 1995). The other (2,000 mg/kg) was measured in a sample collected by Dames & Moore at Station 25D, located near the EFA, and was believed to be associated with sludge management in this area. During the remedial investigation, additional sampling was conducted in this area. These data were included as part of the evaluation of the EFA.

- South Ditch soil/sediment: antimony, arsenic, lead, and cPAH compounds (i.e., benz[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, dibenz[a,h]anthracene, and indeno[1,2,3-cd]pyrene)
- OCA soil/sediment: antimony, arsenic, and lead.

6.2.2.2 Screening Evaluation for Groundwater and Surface Water

Groundwater—A similar screening process was applied to evaluate groundwater concentrations using data collected at the EFA. Specifically, maximum observed Site concentrations of PCoIs in groundwater at the EFA were compared with drinking water standards. For all PCoIs except lead, the Site concentrations were compared with MCLs. Lead concentrations were compared to the national primary drinking water regulation (NPDWR) concentration of 15 μg/L, a standard that is comparable to the MCLs for most chemicals, but which has different monitoring requirements. Data from 15 groundwater samples (five quarters) collected during the remedial investigation were evaluated in this screening. Samples were analyzed for total and dissolved forms of the inorganic PCoIs. Antimony and nickel were not detected in any groundwater samples. As shown in Table 6-9, all detected concentrations were well below the comparison drinking water standards. Based on these results, groundwater was determined to be unlikely to contribute significantly to Site-related risks and was eliminated from further evaluation in the HHRA as an exposure medium of interest.

Surface Water—Samples collected during the remedial investigation from six surface water stations located in the OCA and South Ditch were analyzed for lead (Table 4-25). Total lead was detected in three of six samples, with concentrations ranging from 3.4 to 13.5 μg/L. No lead was detected in samples analyzed for dissolved lead. As with groundwater samples, no observed lead concentration in surface water samples exceeded the NPDWR concentration of 15 μg/L. Based on these results, surface water was determined to be unlikely to contribute significantly to Site-related risks and was eliminated from further evaluation in the HHRA as an exposure medium of interest.

6.3 Exposure Assessment

Exposure assessment includes evaluating: 1) human populations that could potentially come into contact with the CoIs; 2) the environmental transport and fate pathways of those chemicals; and 3) the magnitude, frequency, duration, and route(s) of potential exposure. This subtask of the HHRA builds upon historical and remedial investigation Site evaluations and includes the following components: characterizing the exposure setting (including identifying potentially exposed populations), identifying exposure pathways, defining exposure scenarios, and quantifying exposure.

6.3.1 Characterizing the Exposure Setting

According to EPA guidance (U.S. EPA 1989), the first step in evaluating exposures at a site is to characterize the site's physical features, current and potential future land uses for the site, and

human populations on or near the site. This information is used to identify possible exposure pathways for potentially exposed populations and to determine appropriate exposure intake variables to quantify exposure.

This portion of the risk assessment builds upon work performed in other portions of the remedial investigation, including the approved CSM (PTI and BBL 1997), extracting information necessary for the HHRA. For example, Site evaluations performed to date and data collected during the remedial investigation have characterized the Site history (Section 1.3, *Operational and Regulatory History*) and various aspects of the physical setting (Section 3, *Physical Characteristics of the Site*) that are relevant to the HHRA. Physical characteristics of the Site, such as soil and sediment types, hydrogeology, surface water hydrology, climate, and meteorology, influence the nature of the exposures that human receptors might experience.

Based on Site data, land-use information (as summarized in previous sections), and specific requests from OEPA, the following populations are the primary human receptors with potential exposure to Site-related chemicals:

- Current onsite[4]:
  - Workers, including Thomson employees or contractors performing activities such as periodic NPDES monitoring, inspections, or maintenance in the South Ditch or having occasional contact with soils in other areas of the Site (e.g., the EFA or East Swale)
  - Trespassers contacting soils in the South Ditch
- Hypothetical future onsite, as requested by OEPA
  - Workers contacting soils in the developed portion of the Site (EFA, East Swale, South Ditch)
  - Trespassers contacting soils in the South Ditch
- Current offsite:
  - Recreational users of the OCA (i.e., older children and adults), equivalent to the trespasser potential exposure scenario
- Future offsite:
  - Hypothetical residential use of the OCA (see Appendix I).

In both of the current scenarios, the potentially exposed populations include only adults and older children. Controls on access to the Site and the difficulties of access to the OCA make it

[4] When the initial version of the remedial investigation report was prepared in October 2004, the Thomson facility was still in operation. In the interim period between October 2004 and the current revision of the remedial investigation, the facility has closed, and there are no employees present at the Site. However, the terms "current onsite" and "current offsite" scenarios are used for consistency with the earlier version of this remedial investigation report.

highly unlikely that any young children (i.e., younger than 6 years of age) would contact Site-related materials. The distance and/or physical barriers (e.g., fences, dense vegetation, steep slopes, and/or highway) between any residential areas and the Site and OCA also limit the likelihood that young children would enter the Site or OCA. The exposure assumptions and modeling approaches used to evaluate older age groups are described in Section 6.3.4, *Quantifying Exposures*. The evaluation of a hypothetical residential scenario includes exposures for a young child (i.e., younger than 6 years of age), and exposures for an older child (9–18 years of age). The associated assumptions and modeling approaches are described in Appendix I.

The potential for other populations to be exposed is lower or nonexistent because of the absence of any potentially significant exposure pathway. For example, as documented in Section 6.2.2, *Identifying Chemicals and Areas of Interest*, chemical concentrations in soil samples collected from the adjacent agricultural fields (both on and off the Site) were well below concentrations of potential concern from a risk assessment perspective. As a result, agricultural workers in the Adjacent Fields or individuals consuming edible crops grown on those fields would not have significant exposures to Site-related CoIs. For this reason, these potential receptor populations and associated exposure pathways were excluded from further evaluation in the risk analyses.

Similarly, as discussed below and supported by the data and analyses presented in Sections 4 (*Nature and Extent of Contamination*) and 5 (*Transport and Fate of PCoIs*), the potential for exposure of other current offsite receptors (e.g., nearby residents) is limited because, other than particulate transport to the deltaic area below the railroad tracks in the OCA, negligible offsite migration of Site-related chemicals has occurred through such mechanisms as groundwater transport. The negligible concentrations of CoIs in soil samples collected from most of the Site area limit the potential for significant offsite transport of CoIs to nearby receptor populations (e.g., nearby residents) via windborne fugitive dust emissions. Similarly, as discussed in Section 5, because of the negligible impacts of the Site on underlying groundwater, it is also unlikely that onsite or offsite users of local groundwater would be exposed to Site-related CoIs through groundwater use. Therefore, this potential receptor population and potential exposures to Site-related CoIs associated with inhalation of airborne CoIs or with consumption or other uses of groundwater were excluded from further evaluation in the risk analyses.

Other potentially exposed populations were excluded from quantitative risk evaluations because of their limited potential degree of exposure to Site-related CoIs in certain areas. For example, trespassers could potentially enter the Thomson property; however, such entry is discouraged by access limitations (e.g., fencing). As a result, the degree and frequency of exposure of these individuals to onsite materials would be insignificant, and certainly less than that for onsite workers. However, at the request of OEPA, potential exposures by a trespasser contacting soils or sediments in the South Ditch were evaluated. Similarly, individuals may occasionally trespass on the agricultural fields in the vicinity of the Site. As noted above, however, concentrations of Site-related CoIs in these fields indicate no potential health concern for agricultural workers. Because the degree and frequency of exposure for occasional trespassers would be less than that for agricultural workers, the concentrations observed in the Adjacent Fields do not pose a potential health concern for these individuals, and this exposure scenario was excluded from the quantitative analysis in the HHRA.

Future potentially exposed populations are likely to be similar to those considered for current exposure scenarios. Although Thomson ceased manufacturing operations in 2004, anticipated future uses for the prior industrialized portion of the property will remain consistent with the current zoning regulations, restricting uses to commercial/industrial activities. However, at the request of OEPA, to address hypothetical future exposures by a worker, additional exposure frequencies were added to the quantitative assessment, as described in detail in Section 6.3.4.

In the areas outside the former Thomson facility boundaries, the land use change by the construction of the Wal-Mart Super Center Store or any potential land use changes are unlikely to result in substantially different exposure or risk levels. For example, as discussed in Section 6.2.2, *Identifying Chemicals and Areas of Interest*, chemical concentrations measured in soil samples collected from the adjacent agricultural fields were less than conservative risk-based screening concentrations or were typical of background concentrations. In particular, as discussed with OEPA (2009a), the soils data from the Adjacent Fields was determined to provide appropriate background arsenic concentrations for this Site. The screening concentrations that were used included values calculated assuming standard target acceptable risk levels and residential land use. Therefore, exposure and risk levels associated with Site-related CoIs in these fields would not exceed acceptable target risk levels or risk levels comparable to those associated with background concentrations, even if residential land use were to occur in these areas in the future.

The OCA riparian corridor is largely surrounded by undeveloped lands, particularly the region to the west of the railroad tracks where the offsite creek drainage drops abruptly to the floodplain of the Scioto River. The terraced upland area west of Highway 23 and north and south of the offsite creek consists of undeveloped fields and some commercially developed lands. The westernmost upland area north of the offsite creek also includes some lands proposed for future residential development, although such development may be restricted due to the limited access points available to cross the railroad tracks in this area. West of the railroad tracks to the south, the OCA is bordered by undeveloped lands owned by the EHWD. With the exception of a small water treatment plant and associated infrastructure (accessible only from the south), this area consists of open fields and partially forested areas on rolling terraced uplands approximately 40–50 ft above the OCA corridor. A steep embankment separates these terraced uplands from the OCA along its southern border. North of this area, the OCA is bordered by extensive agricultural fields in the low-lying floodplain between the Scioto River (to west and north) and the terraced uplands north of the offsite creek described above.

For the OCA, land uses that would result in more extensive exposure (e.g., residential land use) are unlikely to occur because these areas are within the 100-year flood plain of the Scioto River. As a result, the approved CSM and HHRA work plan approach did not originally include potential risks associated with hypothetical future changes in land use and consequent exposures. After the draft April 1998 remedial investigation report submittal, OEPA requested a hypothetical future residential scenario be evaluated for the OCA, and it was agreed that the analysis for this hypothetical scenario would be included as an attachment to the remedial investigation report; Appendix I presents this evaluation required by OEPA. The OCA's hypothetical future residential scenario was requested by OEPA because neither GE nor Thomson has control of the future land use and development of the OCA, because it is owned

by a local resident farmer. In 2007, the current property owner executed a Declaration of Use Restriction (DUR) to limit the future use of the contaminated areas of the OCA (GE 2009).

6.3.2 Identifying Exposure Pathways

An exposure pathway is defined as the course a chemical takes from a source to an exposed organism. Exposure pathways include the following four elements: 1) a source; 2) a mechanism of release, retention, or transport of a chemical in a given medium (e.g., air, water, or soil); 3) a point of human contact with the source or affected medium (i.e., an exposure point); and 4) a route of exposure at the point of contact (e.g., ingestion or inhalation). These elements are shown on the conceptual site model (Figure 6-1). If any of these elements is missing, the pathway is considered incomplete (i.e., it does not present a means of exposure). Only those exposure pathways judged to be complete are typically quantified in a risk assessment.

Environmental media to which human receptors could be exposed at the Site include:

- Soil and sludge
- Sediments (including those in the South Ditch and OCA)
- Surface water
- Groundwater
- Air.

Biotic uptake into edible plants, livestock, or game animals is also possible. As discussed below, however, screening evaluations indicated that these secondary media would not contribute significantly to total Site-related exposures.

The primary exposure pathway that was quantitatively evaluated in the risk assessment is incidental ingestion of soil, sediment, and sludge. For the purposes of assessing potential human exposures and risk, no distinction was made between soil and sediment in the Site areas of interest. This approach was taken because of the nature of these materials in the vicinity of the Site. In particular, the degree to which some areas are inundated, particularly in the OCA, varies. In the risk calculations, exposure parameters for soil (e.g., soil ingestion rates) were used to estimate exposures for both soil and sediment. In addition, in the screening evaluations described in Section 6.2.2.1, *Screening Evaluation for Soil, Sediment, and Sludge*, risk-based screening levels based on exposures to soil were used for all comparisons with data for these materials.

Other exposure pathways that were considered in designing the HHRA include:

- Ingestion
  - Ingestion of groundwater used as a drinking water supply
  - Incidental ingestion of surface water
  - Ingestion of crops, livestock, and game
- Inhalation of airborne contaminants
- Dermal contact.

As described below, as a result of the screening level evaluations, these pathways were eliminated from detailed quantitative evaluations in the HHRA.

6.3.2.1 Ingestion of Groundwater

Potential exposures through consumption of or other contact with groundwater are unlikely to contribute significantly, if at all, to total exposures because Site-related PCoIs have not adversely affected Site groundwater and future adverse impacts are unlikely. In particular, as shown in Table 6-9, groundwater monitoring of the shallow aquifer undertaken during the remedial investigation indicated that all detected concentrations of PCoIs were less than federal standards for drinking water supplies.

As discussed in Sections 3.3, *Geology and Soils*, and 3.4, *Hydrogeology*, existing data indicate that a second, deeper aquifer, which is used as a water supply at the Site and by the surrounding region, is hydraulically separated from the shallow groundwater zone. Stratigraphic data indicate that clay layers are present between the shallow and deeper aquifers. The clay unit has been observed to be 20–70 ft deep at locations beneath the Site. Historical monitoring indicates that the deeper aquifer has not been affected by chemicals from the Site (E&E 1989; Keystone 1990). Based on historical and remedial investigation data, human exposure to Site-related chemicals in groundwater is unlikely. For these reasons, groundwater was excluded from the quantitative risk evaluations as a potential pathway for exposure to Site-related CoIs.

6.3.2.2 Incidental Ingestion of Surface Water

Site information indicates that the potential for human exposures to Site-related CoIs in surface water is negligible. On the Scioto River, which ultimately receives discharges from the OCA, there are no drinking water intakes within 3 miles of the Site (E&E 1989). Surface water exposures would consist only of incidental ingestion of surface water in the OCA or South Ditch during recreation or other activities involving water contact. Because such exposures are likely to be small in magnitude and infrequent, surface water exposures are unlikely to contribute significantly to total exposures. In addition, during the remedial investigation, surface water at various locations in the South Ditch and other downstream locations was monitored for lead. As discussed in Section 6.2.2.2, *Screening Evaluation for Groundwater and Surface Water*, all

dissolved lead concentrations and most total lead concentrations were less than the detection limit. All lead results were less than the drinking water criterion for lead (15 μg/L).

In addition, existing sampling data and geochemical investigations indicate that PCoIs are strongly bound to the sediments where they occur. Therefore, dissolution of PCoIs from sediment, which could affect surface water quality, is not expected to occur to any significant degree. For these reasons, surface water was excluded from the quantitative risk evaluations as a potential pathway for exposure to Site-related CoIs.

6.3.2.3 Ingestion of Crops, Livestock, and Game

Potential human exposure pathways reflecting biotic uptake of CoIs into food sources such as crops, livestock, and game reflect more attenuated, secondary human exposure pathways than such pathways as direct ingestion of drinking water or soil (i.e., several additional steps are required before chemicals in water or soil would reach human receptors). Such pathways require that the CoI be taken up by the food species (e.g., from water or soil) and that the food species then be consumed by human receptors. Both the nature of the CoIs at the Site and the Site data and conditions suggest that the contributions of environmental media involving food sources (e.g., game or edible crops) to overall exposures and risk are minimal.

First, the biotic uptake of metals such as lead is generally low and such metals tend not to accumulate to a significant degree in plant or animal tissues (U.S. EPA 1986; Chaney and Ryan 1994). This factor would reduce the potential for Site-related CoIs to be transferred to human receptors via biotic uptake. Second, for agricultural products that might be raised in the vicinity of the Site, as discussed in Section 6.2.2.1, *Screening Evaluation for Soil, Sediment, and Sludge*, the concentrations of CoIs that were observed in the agricultural fields in the vicinity of the Site were low and were less than risk-based screening concentrations derived assuming direct ingestion of the soil in a residential scenario. Because these concentrations are not associated with any significant risk, even if the soil were to be ingested, risks associated with more attenuated exposure pathways would be less. Thus, consumption of crops or livestock raised on these fields would present a negligible exposure potential. The third factor supporting the negligible contributions of these biotic uptake pathways to overall Site-related exposures is the small proportion of the total diet that might be obtained from potentially affected food sources. In particular, game that might be obtained from the OCA is unlikely to constitute a substantial component of the food supply of any individual. This factor would again serve to reduce the potential contribution of these exposure pathways to overall Site-related exposures.

6.3.2.4 Inhalation of Airborne Contaminants

Air pathways are negligible contributors to Site-related exposures and risks. Ambient air monitoring has shown that the air in the vicinity of the Site is in full compliance with federal standards for ambient air lead concentrations and that the Thomson plant met its EPA air permit lead emissions limits. The potential for significant human exposures for onsite workers or nearby residents to fugitive dust emissions is also negligible because of factors that reduce the potential for CoIs in soil or sediment to become airborne (e.g., the presence of a clean topsoil cap in the EFA and the presence of water or vegetative coverings in the South Ditch and OCA).

The low concentrations of CoIs in soil over much of the Site and surrounding areas also reduce the potential for significant exposures to occur via fugitive dust.

In addition to inhalation of airborne dust, inhalation of volatile chemicals can also be a potential exposure pathway of concern at sites where such chemicals are present. The data collected during the remedial investigation, however, confirm that VOCs are not CoIs for the Site and that inhalation of VOCs is not a concern. Specifically, in accordance with the remedial investigation work plan, soil samples from the Former Oil Skimmer Pond and sediment samples from the South Ditch were collected and analyzed for a full suite of VOCs. Only three VOCs were detected in soil samples collected from the Site. Each compound was detected in only one sample at insignificant concentrations. As shown in Table 6-4, the detected concentrations were all well below risk-based screening levels developed by EPA for a residential exposure scenario. Thus, this exposure pathway was excluded from the quantitative risk analyses.

6.3.2.5 Dermal Contact

Dermal contact and consequent absorption were also excluded from the quantitative evaluations because of the nature of the CoIs at the Site. Most of the CoIs are metals. Dermal absorption is generally an insignificant exposure pathway for metals because metals are typically poorly absorbed through the skin (Ryan et al. 1987). In particular, for lead, the primary CoI at the Site, dermal exposure is not included in the chemical-specific exposure assessment approaches for this metal because of the negligible contributions of this pathway to overall lead exposures. The only organic chemicals identified as CoIs for the Site are certain cPAH compounds that were observed in soil/sediment samples collected from portions of the South Ditch.

Because carcinogenic PAH compounds are present at the Site only to a limited extent, omitting this potential exposure pathway from the risk evaluations is unlikely to significantly affect the risk assessment results. Based on these considerations, dermal exposure to cPAH compounds and metals was excluded from the quantitative risk analyses.

Other factors related to the nature of the exposures that would be expected at the Site also support the conclusion that omitting the dermal exposure pathway from quantitative risk calculations will not cause the HHRA to overlook any significant sources of risk or exposure associated with the Site. As discussed in more detail in Section 6.3.4, the types of exposures to materials in the OCA and onsite areas that were evaluated in the quantitative risk analyses are likely to be infrequent and of short duration. Moreover, as described in Section 6.5, estimated exposures and risks for more direct exposure pathways (e.g., incidental ingestion of soil, sludge, or sediment) were within acceptable levels and were generally significantly less than levels that would be of concern. Excluding exposure and risk estimates for the substantially smaller magnitude dermal exposure pathway would not change the overall conclusions of the HHRA or the Site remediation decisions made based on those results. Additional discussion regarding exclusion of the dermal pathway is discussed in Section 6.6.2.3, *Uncertainties in Exposure Pathways*.

6.3.3 Defining Exposure Scenarios

Following the screening processes described in Section 6.2.2, *Identifying Chemicals and Areas of Interest*, Section 6.3.1, *Characterizing the Exposure Setting*, and Section 6.3.2, *Identifying Exposure Pathways*, the remaining potential exposure scenarios and pathways of interest for each Site area were compiled. The results of the screening process are summarized in Table 6-10, while the exposure scenarios and pathways evaluated quantitatively for each Site area are summarized in Table 6-11. Exposure scenarios for which quantitative analyses were performed for the HHRA include the current and hypothetical future onsite occupational scenario, the trespasser scenario, and the current offsite recreational scenario; as noted previously, a hypothetical future residential exposure scenario was completed for the OCA and this analysis is presented in Appendix I. As described in Section 6.3.1, *Characterizing the Exposure Setting*, the exposure scenario for agricultural workers shown on Figure 6-1 was excluded from further quantitative analyses because the concentrations of CoIs measured in soil samples collected from the agricultural fields in the vicinity of the Site were less than risk-based screening levels.

Assumptions regarding exposure scenarios and pathways were combined with chemical concentration data gathered in the remedial investigation to generate intake levels and, ultimately, to quantify risks. Table 6-11 summarizes the Site media that were evaluated in each exposure scenario.

6.3.4 Quantifying Exposures

For lead, chemical-specific models were used to estimate exposure as reflected in predicted blood lead concentrations. Because of the ubiquity of lead in the environment from multiple sources, lead exposure models implicitly or explicitly incorporate contributions from a variety of potential sources of lead, including sources that are not Site-related. For other CoIs, daily chemical intake rates were estimated for the exposure pathways identified during the screening process described in Section 6.3.2, *Identifying Exposure Pathways*, as being the most significant contributors to total exposures and risks (i.e., incidental ingestion of soil and sediment). These estimated exposure levels and intakes for lead and other CoIs were calculated from EPCs (i.e., concentrations in environmental media at the locations where exposures could occur) and from the estimated magnitude of exposures to media containing CoIs. The specific approaches used to quantify exposures to lead and other CoIs are described below.

Data generated during the remedial and supplemental investigations were reviewed together with historical data to develop appropriate EPCs. The distribution of the available concentration data and appropriate summary statistics for those data were considered in selecting appropriate EPCs for use in the risk assessment. Sources of EPC data were as follows:

- Measured onsite and offsite concentrations in soil and sediment
- Measured concentrations in sludge.

As a conservative element of these analyses, exposure estimates for individual exposure areas were combined, where such combinations are reasonable, to develop estimates of total exposure

associated with multiple exposure sources. EPCs were developed for each Site area of interest examined in the quantitative risk analyses undertaken in the HHRA. EPCs were also developed for selected, more focused subareas of interest, e.g., in the vicinity of outfalls along the South Ditch where periodic NPDES monitoring occurs or has occurred.

As discussed in Section 6.3.2, *Identifying Exposure Pathways*, for the purposes of the human health risk assessment, no distinction is being made between soil and sediments. This approach is being taken because it is difficult to clearly distinguish between soil and sediment in some areas and because the distinction between soil and sediments is unlikely to significantly affect human exposure potential for these materials. The exposure algorithms for the onsite worker scenario presented in this section were also used to evaluate exposures to sludge in the EFA.

Geographic considerations played a role in appropriately quantifying exposures. Likely types and frequencies of exposures as well as access and proximity to specific locations were considered. For example, hindrances to access to the OCA were considered when quantifying likely exposure frequencies for this area. As noted earlier, a hypothetical future residential scenario was evaluated assuming potential contact up to 350 days/year (Appendix I).

For lead, exposure estimates are expressed as the lead concentration in blood that is associated with specified conditions of exposure. For other CoIs, exposure estimates for ingestion, or "intakes," are defined as the mass of a chemical taken into the body, per unit of body weight, per unit of time. EPA recommendations regarding appropriate approaches for assessing lead exposures and risks (U.S. EPA 1994d, 2003a) were applied in these analyses. These models generate a distribution of predicted blood lead concentrations or a specific point on that distribution. For other CoIs, chemical intakes for chronic exposures, or chronic daily intakes (CDIs), were calculated for specific exposure media by the methods described in U.S. EPA (1989) and using exposure parameters presented in relevant EPA guidance (U.S. EPA 1991a, 1997, 2008). Consistent with this guidance, exposure estimates for other CoIs were designed to reflect what EPA defines as the reasonable maximum exposure (RME) case, an estimate that EPA states is intended to provide a risk estimate that is conservative yet is within the range of possible exposures at a site.

Only slight modifications or additions to the generic guidance for lead and other CoIs were made in performing the HHRA. In particular, because EPA guidance regarding recreational/ trespasser scenarios is limited, the selection of approaches and exposure parameters for risk calculations for this scenario was based on consideration of Site-specific conditions and characteristics and on protection of public health. Assumptions regarding exposure frequencies for the current onsite worker scenario were also modified to reflect Site characteristics and conditions of Site use, with additional consideration of hypothetical future workers and trespassers included at the request of OEPA.

All exposure algorithms and the associated exposure parameters and assumptions used to calculate lead exposure estimates or CDIs for each of the exposure pathways listed above are documented in the subsections that follow.

6.3.4.1 Quantifying Exposures for Lead

In contrast to risk assessment techniques for most other chemicals, toxic effects of lead are typically correlated with observed or predicted biological indices, e.g., blood lead concentrations, rather than with calculated exposure or intake levels. Because of certain features of lead toxicology, pharmacokinetics, and exposure data, efforts to assess the potential risks associated with lead exposures use a different methodology than is typically used in quantifying risks associated with exposures to other chemicals. Specifically, instead of calculating incremental cancer risks or hazard indices based on quantitative estimates of chemical intake from Site-related exposure sources, lead risks are assessed by predicting the body burden of lead (as reflected in blood lead concentrations) that will result from assumed intake and uptake of lead from a variety of sources. Such an approach has been taken, in part, because the multiple sources and ubiquity of lead in the environment complicate attributing lead exposures to specific sources or quantifying medium-specific intake rates corresponding to identified health effects.

Because of these differences in assessing lead exposure and toxicity, significant effort has been expended by EPA and the scientific community to develop alternative methods to quantify the potential adverse health effects of lead. In particular, models have been developed to predict the contributions of various environmental exposures to blood lead concentrations, just as such exposures are typically used to quantify chemical intake rates for most other chemicals. EPA currently recommends a model for use in evaluating the lead exposures of young children (less than 6 years old) in a residential setting (U.S. EPA 1994d, 2009a; i.e., the integrated exposure uptake biokinetic [IEUBK] model). The IEUBK model incorporates assumptions regarding lead exposure from a variety of sources, and estimates blood lead concentrations based on a detailed pharmacokinetic model of lead distribution in the human body. For adults, EPA developed a simplified adult lead model (ALM) in 1996 as an interim approach to assessing adult exposure to lead in soil, with the intent of developing a model similar to IEUBK at a later date (U.S. EPA 2003a). In 2001, an EPA review of the ALM and other published models used to assess nonresidential adult exposures to lead concluded that no other models offered significant improvements to the ALM, and recommended continued use of the existing model (U.S. EPA 2001, 2003a). EPA is currently in the process of developing an all-ages model that may provide improvements to the ALM (U.S. EPA 2001). The ALM was applied in the evaluations of the Site for older children and adults (as discussed below). For hypothetical future residential exposures, the IEUBK model was used to assess exposures for a young child, and the ALM was used to assess exposures for an older child (Appendix I).

The algorithm used to estimate adult lead exposures is shown in Table 6-12. This model uses a biokinetic slope factor to predict blood lead concentrations associated with specified lead concentrations in soil and incorporates consideration of background blood lead levels arising from other exposure sources, predominantly dietary lead. The model also employs assumptions regarding soil ingestion rates, absorption of lead from soil, and the frequency with which such exposures occur. Central tendency estimates of the various input parameter estimates are typically applied in the model to yield an estimate of the geometric mean blood lead concentration in the target population. The potential variability in blood lead concentrations is reflected by use of a geometric standard deviation (GSD), as described below, to generate a distribution of predicted blood lead concentrations or a specific point on the distribution,

(e.g., the 95th percentile estimate of the blood lead concentration in an exposed target receptor population [$PbB_{target,0.95}$]). Typically, the receptor is identified as a fetus whose mother is exposed to lead in soil under the assumed conditions. As described in Section 6.5, *Risk Characterization*, the model result is then compared with a selected target level to assess the potential for adverse health effects associated with lead exposures at the Site. Specific assumptions used in the modeling for the Site are discussed below.

The ALM was designed primarily to evaluate potential lead exposures of adults in a workplace setting and is most appropriately applied where exposures are regular (e.g., occurring daily or several times per week). As discussed below, the types of exposures to the Site areas of interest are typically intermittent, infrequent, and of limited duration. However, the ALM assumes a quasi-steady state for blood-lead levels and thus cannot be used to model exposures that occur less than once per week (U.S. EPA 2003a). Therefore, although the most realistic exposure frequency is likely to be significantly less than once per week, the minimum exposure frequency supported by the ALM of once per week was used in estimating exposures at the Site, along with higher exposure frequencies. The implications of this are discussed in the uncertainty analysis section of this report, Section 6.6.2, *Uncertainties Associated with Exposure Assessment.*

In addition, in the recreational user/trespasser scenario, the assumed receptor population includes older children (9–18 years of age) as well as adults. No standard default assumptions exist for evaluating potential lead exposures in older children, so a "matrix" approach, combining varying exposure frequencies and soil ingestion rates, was used in order to provide results from a full range of potential exposure scenarios. Use of some of the elements of the ALM will provide conservative estimates of exposures in this population, while others may underestimate blood lead concentrations for this group. Most significantly, use of a target blood lead concentration based on fetal exposures is highly health-protective. This level is designed for protection of fetuses, as well as young children (0–6 years of age), who are more vulnerable to the adverse health effects of lead. In addition, younger members of the age group examined in the recreational/trespasser scenario (i.e., children between 9 and 18 years of age) who would be exposed in the South Ditch and OCA are unlikely to be pregnant. Thus, for the exposure assessment for the recreational user/trespasser scenarios, calculations were performed for two receptors. First, the model was applied using a fetus as the target receptor of concern. A second set of calculations was performed without the ratio between the fetal and maternal blood lead concentrations ($R_{fetal/maternal}$). This change yields the blood lead concentration for the older child, rather than a fetus, and results in higher blood lead concentrations than would be modeled for the fetus. The implications of using the ALM to assess lead exposures in older children are more fully evaluated in the uncertainty analysis section of this report, Section 6.6.2, *Uncertainties Associated with Exposure Assessment.*

As noted earlier, a hypothetical future residential scenario was also evaluated for the OCA. This evaluation was conducted using the IEUBK model, which is consistent with agency guidance for assessing exposures to lead under a residential exposure scenario. Additionally, a scenario was evaluated using the ALM to assess exposures that might occur for an older child (i.e., 9–18 years of age) visiting the deltaic or upper creek areas of the OCA. The input parameters, results, and uncertainties associated with the assessment of the hypothetical future residential scenario are presented in Appendix I.

Baseline Blood Lead Concentration in Adults ($PbB_{adult,0}$)—The baseline blood lead concentration is intended to reflect the blood lead concentration that would be expected in a potentially exposed population from sources other than Site-related sources. Baseline sources include lead in food and water. Where adequate site-specific data are not available, EPA recommends use of baseline values obtained based on U.S. data collected as part of the National Health and Nutrition Examination Study (NHANES). The most recent update to the ALM guidance recommends the use of the most current NHANES results for baseline blood lead concentrations rather than the values presented in the ALM guidance itself (U.S. EPA 2003a). The values from NHANES reflect nationwide averages calculated for women of child-bearing age. EPA recently updated their recommendations for these parameters for adults, and now recommends using a geometric mean of 1.0 μg/dL for all applications of the ALM, based on NHANES III and subsequent NHANES surveys (1999–2004) (U.S. EPA 2009b). EPA no longer recommends segregating data by geographic region or ethnicity because of the lack of homogeneity of populations and the potential for receptors to move between regions. Thus, this assessment used the EPA-recommended geometric mean blood lead value of 1.0 μg/dL for the adult worker scenarios. For the trespasser scenarios, there are no EPA recommendations for children ages 9–18 years; so following direction from OEPA (2009b), the baseline blood lead value for adults was also used for the older child exposure scenarios.

Soil Lead Concentration (PbS)—Based on the structure of the lead model and EPA practice in applying lead models, an estimate of the central tendency for Site concentrations is needed as an input. A geometric mean lead concentrations in soil, sediment, and/or sludge (in mg/kg) was originally selected in the draft remedial investigation report as the most appropriate exposure concentration because the data set for this Site is likely to be biased high (i.e., areas for which concentrations were expected to be higher were sampled more often), and because complete environmental data sets are typically lognormal (U.S. EPA 1992b). However, OEPA requested that calculations use the arithmetic mean for the central tendency estimate. Therefore, all modeling for lead in this report has been revised to use an arithmetic mean rather than a geometric mean. Arithmetic means, when compared to geometric means, are more significantly affected by a small number of high values. Thus, exposures at this Site estimated using an arithmetic mean are likely to overestimate actual exposures. It should also be noted that the input parameter required by both the ALM and the IEUBK model for children is a mean concentration rather than an upper confidence limit (UCL) on the mean. EPA's most recent guidance on calculating UCLs for exposure point concentrations at hazardous waste sites specifically directs that a mean or average concentration, rather than an UCL on the mean, be used for evaluations of lead (U.S. EPA 2002b). This directive also appears on the online frequently asked questions (FAQs) for the ALM (U.S. EPA 2009c), and a "short sheet" for the IEUBK model (U.S. EPA 2007). Therefore, arithmetic mean lead concentrations were calculated for each Site area of interest using the entire Site area as the assumed exposure area. In addition, concentrations were derived for certain specific subareas with special exposure potential (e.g., in the vicinity of outfalls to the South Ditch where periodic NPDES monitoring occurs or has occurred). When calculating EPCs, the distribution of the available historical data, and all remedial investigation data within a specific Site subarea of interest was considered.

An average lead concentration was calculated using data from all depths, then depth-specific concentrations were calculated for every 6-in. interval up to a depth of 2 ft. The evaluation of

depth-specific soil concentrations was intended to encompass potential future scenarios that might allow deeper soils to reach the surface (e.g., septic system repair, gardening) (U.S. EPA 2003b), as specifically requested by OEPA. Results using all these EPCs are presented in the tables (Tables 6-16, 6-18, and 6-19), with the maximum average soil concentration for each exposure area highlighted in bold font. The basis for these values is provided in Appendix F. All information regarding the hypothetical future residential scenario is presented in Appendix I.

Biokinetic Slope Factor (BKSF)—Biokinetic slope factors are empirically derived values that relate blood lead concentrations to lead uptake or lead concentrations in a specific environmental medium. The slope factor recommended by EPA for use in the simplified adult model is expressed in units of blood lead concentration increment per increment of lead uptake into the body (e.g., μg/dL blood lead increase per μg/day lead uptake). EPA recommends a value of 0.4 for this parameter based on data relating tap water concentrations and blood lead concentrations in adult males (U.S. EPA 2003a). As discussed in Section 6.6.2.2, based on information from EPA's IEUBK model for children (U.S. EPA 2009a), use of this parameter is likely to be conservative for scenarios involving both adults and older children.

Soil Ingestion Rate (IR_s)—As discussed above, exposures to lead were evaluated for adults, older children, and younger children. For younger children, the default soil ingestion rates recommended for use in the IEUBK model were used, and are discussed in Appendix I. For adults and older children, a range of soil ingestion rates was used. For one set of calculations, the default soil ingestion rate of 50 mg/day recommended by EPA for use in the adult model (U.S. EPA 2003a) was assumed for receptors in both exposure scenarios. As discussed later in Section 6.3.4.2, this value is consistent with other agency recommendations regarding soil ingestion.

In addition, for the trespasser scenarios, a lower soil ingestion rate of 25 mg/day was included. This lower soil ingestion rate was based on multiple factors. The default ingestion rate of 50 mg/day is based on occupational exposure to soil and dust, and implicitly assumes a full 8-hour workday, whereas a trespasser is more likely to be in contact with Site-related soils for only a few hours at each visit. Additionally, the default ingestion rate incorporates ingestion of both outdoor soil and indoor dust. However, only a fraction of an individual's total daily soil ingestion comes from outdoor sources. Specifically, the IEUBK model recommends a value of 0.45 as the fraction of the total soil ingestion attributable to outdoor soil rather than indoor dust. The few studies that have attempted to quantify the amount of soil versus dust ingested by children appear to be consistent with EPA's default assumption for outdoor soil ingestion (e.g., 49 percent; Stanek and Calabrese 1992). EPA Region VIII has also used a fractional intake of 0.45 for outdoor soil versus indoor dust ingestion in risk assessments of arsenic in soil when the concentrations of arsenic in soil and house dust differ. A soil ingestion rate of 25 mg/day is consistent with this understanding of sources for inadvertent soil ingestion, and with the IEUBK analysis conducted for the residential scenario. Even this lower soil ingestion rate is likely to overestimate actual ingestion of soil from the OCA or the South Ditch because only a portion of a child's total time outdoors would be spent visiting this area. It is unlikely that the exposure time for each visit would account for 100 percent of outdoor exposure time on each visit to the area. However, it is difficult to quantify the specific fraction of outdoor time that might be spent in the specific areas.

Finally, this assessment also incorporated a higher soil ingestion rate of 100 mg/day to address OEPA's concerns regarding potential contact-intensive activities under both worker and trespasser scenarios. This value was recommended by the Technical Review Workgroup for Metals and Asbestos as an appropriate value for contact-intensive occupational tasks such as digging or construction work (U.S. EPA 2003c; U.S. EPA 2009c). Because the value of 100 mg/day was derived for a full 8-hour work day applicable to a worker scenario (which is likely to overestimate a trespasser's exposure duration), this soil ingestion rate should provide a high-end exposure estimate for the recreational user/trespasser scenario.

Absorption Fraction of Lead from Soil (AF_s)—As a conservative measure, the default soil lead absorption factor provided in EPA's recommendations was used in the model. This value (0.12) reflects two components. First, the absorption of soluble lead is assumed to be 20 percent (0.2). Second, the absorption of lead from soil is assumed to be 60 percent (0.6) of the absorption of soluble lead. By multiplying these two factors, an absolute absorption fraction for soil lead of 0.12 is derived (i.e., 0.2×0.6). This value was also assumed to apply to lead in sediment and sludge in the Site areas of interest. As discussed in Section 6.6.2, *Uncertainties Associated with Exposure Assessment*, this value represents a conservative estimate of the likely average absorption of lead from soil.

Exposure Frequency (EF_s)—In selecting the assumption for exposure frequency, the actual patterns of contact with the Site areas of interest were considered, as well as hypothetical future scenarios. For both of the exposure scenarios of interest, exposures are likely to be intermittent, infrequent, and limited in duration and magnitude. For example, within the occupational scenario, the Thomson employee or contractor who performs NPDES monitoring or conducts inspections would be potentially exposed to Site-related materials in the South Ditch only at the monitored locations and only during the monitoring events, which occur infrequently (i.e., approximately once per week) and for only a short time (i.e., less than 1 hour). To collect the necessary monitoring samples, contact with ditch sediments is not required. Similarly, access to chemicals in areas such as the EFA or the East Swale is limited by the presence of fences, the fact that these areas are located in controlled access areas, and the presence of a soil cover layer over virtually all of the sludge materials. These areas also are not part of operational activities, so there is generally no need for workers to enter them. Access to the OCA is also limited by its isolated location and difficult public access. In addition, the most elevated concentrations of lead and other CoIs are generally found within only a small subarea of the OCA (i.e., the deltaic area below the railroad tracks where the offsite creek bifurcates before its confluence with the farm ditch). As a result, exposures to these areas are expected to be intermittent and of limited duration and magnitude.

Based on these considerations, an exposure frequency of 1 day per month (12 days/year) is likely to be the most realistic, yet conservative (i.e., health-protective) estimate of the exposure frequency for worker exposure to most of the areas of interest. However, the ALM assumes a quasi-steady state for blood lead and thus cannot be used to model exposures that occur less than once per week (U.S. EPA 2003a). Thus, exposure estimates were calculated assuming weekly exposures (50 days/year) to each area of interest, an assumption that reflects current exposures in the vicinity of the NPDES outfall, but also significantly overestimates exposures based on actual current or likely future patterns of Site use for most Site areas. Additional

exposure frequencies (twice weekly, every other day, and daily) were added to the assessment at the request of OEPA, to provide a range of estimates for hypothetical future exposures.

Similar assumptions were applied for evaluating the South Ditch and OCA in the recreational user/trespasser scenario. Because of the difficulties of access to these areas, an exposure frequency of 1 day per month (12 days/year) is likely to be the most plausible, yet conservative (i.e., health-protective) estimate of the exposure frequency. However, as described above, the ALM assumes a quasi-steady state for blood lead and thus cannot be used to model exposures that occur less than once per week (U.S. EPA 2003a). Thus, weekly exposures (50 days/year) were evaluated for recreational users/trespassers. As indicated above, additional exposure frequencies (twice weekly, every other day, and daily) were added to the assessment at the request of OEPA, to provide a range of estimates for hypothetical future exposures. These higher exposure frequencies are likely to overestimate exposures because, on a given day of exposure to the South Ditch or OCA, the receptor is likely to have exposures to soil in areas outside these areas that will not contain elevated lead concentrations. In addition, a receptor will only contact soil with the assumed EPC for a portion of the time. The overall sampling for the two areas focused on areas where elevated lead concentrations were expected to occur. Thus, the EPC developed based on these data is biased toward higher concentrations than would be encountered, on average. This factor is particularly important for those analyses undertaken using the EPC developed for the subsets of the South Ditch or OCA data where the highest lead concentrations were observed, as these areas constitute only a small portion of the total area of interest.

Averaging Time (AT)—The averaging time is the total period of time over which soil contact might occur. This value was assumed to be 365 days/year in these evaluations.

Geometric Standard Deviation (GSD)—The GSD is intended to reflect individual variability in blood lead concentrations that might result from a specified degree of exposure. This variability among exposed individuals arises from such sources as biological variability (e.g., differences in lead absorption) and behavioral variability (e.g., differences in soil ingestion rates and patterns of soil ingestion relative to food ingestion). The GSD is used to calculate the blood lead concentration corresponding to a specified percentile (e.g., the 95th percentile) of the probability distribution of blood lead concentrations for an exposed individual.

Few data are available regarding GSD values reflecting individual variability. Instead, GSD values reflecting community variability are frequently applied to estimate individual variability. GSDs based on community variability are greater than those reflecting individual variability because community GSDs incorporate additional sources of variability such as differences in exposure sources among individuals (e.g., differences in lead concentrations in drinking water). As a result, use of community GSDs to represent individual variability will tend to overestimate predicted blood lead concentrations at various percentiles. Lower GSD values reflect lower variability in exposure conditions and population characteristics.

In the absence of adequate Site-specific blood lead concentration data, EPA recommends the use of the appropriate value from the most recent NHANES (U.S. EPA 2003a). As described in the section above on the baseline blood lead, based on the most recent values recommended by EPA for the ALM, a GSD of 1.8 was selected (U.S. EPA 2009b). EPA no longer recommends

segregating data by geographic region or ethnicity because of the lack of homogeneity of populations and the potential for receptors to move between regions. Thus, this assessment used the EPA-recommended GSD of 1.8 for the adult worker scenarios. For the trespasser scenarios, there are no EPA recommendations for children ages 9–18 years, so following the advice of OEPA, the above-referenced GSD for adults was also used for the older child scenarios (OEPA 2009c).

Several features of the Site suggest that the potentially exposed population is likely to have relatively homogeneous exposures. First, the Site is located in an area that is not highly urbanized. In addition, there is no information indicating significant sources of lead exposure to the local population (e.g., the presence of a smelter in the community). As discussed in Section 6.6.2, *Uncertainties Associated with Exposure Assessment*, the selected GSD value is likely to represent a conservative estimate of the likely variability in blood lead concentrations in the population in the vicinity of the Site.

Ratio Between Fetal and Maternal Blood Lead Concentrations ($R_{fetal/maternal}$)—Because the adult model is based on exposures by an adult, this ratio is used to account for the slight observed difference between umbilical cord blood lead concentrations and maternal blood lead concentrations. Based on a number of studies in which maternal and fetal blood lead concentrations were compared, the default value for this parameter selected by EPA for fetal blood lead concentration is approximately 90 percent of maternal concentration (U.S. EPA 2003a). As noted above, this value was omitted from certain calculations performed for the recreational user/trespasser scenarios (i.e., to calculate exposures for older children). When this value is omitted, the resulting blood lead concentration is the predicted concentration in the exposed individual (e.g., the older child) rather than in a fetus.

These assumptions were applied in the adult model to predict the blood lead concentrations that would be associated with exposures to lead in soil, sediment, and/or sludge in the Site areas of interest.

6.3.4.2 Quantifying Exposures for Other Cols

The exposure algorithm for ingestion of soil and sediment is presented in Table 6-13. This pathway was examined in both the current and hypothetical onsite occupational scenario and the current and hypothetical recreational user scenarios. This algorithm is based on incidental ingestion of soil or sediment as a result of direct contact with contaminated soil or sediment on hands, followed by hand-to-mouth activity (either inadvertent or associated with eating or smoking).

The concentration term in the exposure equation is intended to reflect the average concentration contacted at the exposure point or points over the exposure period (U.S. EPA 1989). In most situations, assuming long-term contact with the maximum concentration in any exposure medium is not reasonable. EPA guidance (U.S. EPA 1989) states that although the average concentration does not reflect the maximum concentration that could be contacted at any one time, it is regarded as a reasonable estimate of the concentration likely to be contacted over time (U.S. EPA 1989). As recommended by EPA (U.S. EPA 1989), the 95-percent UCL of the arithmetic mean was used in estimating EPCs in all cases for Cols other than lead, because of

the uncertainty associated with estimating the average exposure concentration. UCLs were determined using the most current version of EPA's ProUCL software (Version 4.00.04), which incorporates specialized statistics to account for non-detects. Where the calculated 95-percent UCL on the arithmetic mean exceeded the maximum concentration of the CoI in the medium of interest observed at the Site, the maximum concentration was used as the EPC. As noted above, EPCs were developed for each Site area of interest as a whole. EPCs were also calculated for selected Site subareas (e.g., in the vicinity of outfalls along the South Ditch where periodic NPDES monitoring occurs or has occurred). The data and calculations applied to derive EPCs are shown in Appendix F.

For calculating exposure for a typical industrial worker, older EPA (U.S. EPA 1991a) guidance recommends using a soil ingestion rate of 50 mg/day. Additionally, EPA's Exposure Factors Handbook (1997) states that the value of 50 mg/day for industrial settings are within the range of the three limited studies available for adult soil ingestion, and represents a reasonable central tendency estimate (U.S. EPA 1997). For the recreational/trespasser scenario, a higher value of 100 mg/day was assumed for the daily soil ingestion rate for older children and adults (U.S. EPA 1989, 1997, 2008).

As discussed above in the description of the exposure parameters used in the lead exposure model, the actual patterns of contact with the Site areas of interest were considered when selecting the assumptions for fractional intake and exposure frequency. Because exposures in both of the exposure scenarios of interest are likely to be infrequent and limited in duration and magnitude, an exposure frequency of 1 day per month (12 days/year) was assumed as the most plausible, yet conservative (i.e., health-protective) estimate of the exposure frequency for worker exposure to most of the Site areas of interest. As a comparison, exposure and risk estimates were also calculated assuming weekly exposures to each area of interest (50 days/year). This value reflects the exposure frequency in the vicinity of the NPDES outfall on the South Ditch, but significantly overestimates exposure to other Site areas of interest. As discussed with the OEPA, a fractional intake value of 0.5 was assumed for the recreational user/trespasser scenarios for non-lead exposures. However, at the request of OEPA, a fractional intake of 1 was assumed for the onsite worker.

Worker exposures were assumed to occur over a 25-year period, the standard EPA default value (U.S. EPA 1991a). Receptors in the recreational scenario were assumed to have exposures to soil and sediments in the South Ditch or OCA over a 9-year period, between the ages of 9 and 18 years. This age range was selected for this scenario because this was the age range judged most likely to visit these areas. The body weight for adult workers was assumed to be 70 kg, consistent with EPA's default value for the average adult (U.S. EPA 1991a). A body weight of 49 kg was assumed for the recreational users, based on data for older children (U.S. EPA 2008).

The averaging time used to determine a CDI depends on the type of toxic effect being assessed. For assessing carcinogenic effects, CDIs are calculated by prorating the total cumulative dose over a lifetime, estimated to be 70 years (U.S. EPA 1991a). For assessing noncancer effects, CDIs are calculated by averaging intakes only over the period of exposure. The distinction between these two approaches is based on the currently held scientific opinion that the toxicological mechanisms of action are different for carcinogenic and noncarcinogenic processes. As discussed in Section 6.4, *Toxicity Assessment*, current toxicological theory and

regulatory policy assume that, in the absence of evidence to the contrary, there is no dose for carcinogenic substances that does not present some incremental increase in cancer risk. Therefore, the measure of exposure used in risk calculations is a lifetime-averaged daily dose. By contrast, for noncancer health effects, a threshold dose level is assumed to exist. At exposures less than the threshold, no risk of adverse health effects is assumed to exist. As a result, the measure of exposure used in risk calculations for noncancer health effects is a daily intake during the period of exposure (U.S. EPA 1989).

6.3.5 Summary of Exposure Assessment

The results of the exposure assessment (estimates of EPCs and CDIs) are summarized in the risk characterization tables cited in Section 6.5. As shown in these tables, the CDIs calculated in this step of the risk assessment are combined with chemical-specific toxicity values (described in Section 6.4) to characterize potential risks. The blood lead concentrations predicted by the lead model are compared with a specified target blood lead concentration to assess potential health risks.

Several sources of uncertainty in the exposure assessment affect, to varying degrees, the estimates of exposure and, consequently, the final estimates of risk. In all exposure assessments, there are uncertainties associated with the sampling data (i.e., how well the data represent actual Site conditions) and the exposure assumptions (i.e., how well the assumptions reflect actual or future exposure conditions). In addition, the degree to which the underlying assumptions in the model used adequately reflect actual exposure conditions and characteristics of the potentially exposed population also contribute to uncertainties in the risk assessment results. Because of these uncertainties, conservative intake variables, which are likely to overestimate true exposure, were used to calculate chemical intakes for individual exposure pathways. A more detailed discussion of uncertainties associated with exposure assessment is presented in Section 6.6, *Uncertainty Analysis*.

6.4 Toxicity Assessment

The purpose of the toxicity assessment is to evaluate the potential for Site-related chemicals to cause adverse health effects in exposed individuals and to define, to the extent possible, the relationship between the degree of exposure to a hazardous chemical and the likelihood and severity of any adverse health effects. Standard EPA procedures for assessing and quantifying the toxicity of the CoIs were applied in the HHRA. These procedures include identifying toxicity values for carcinogenic and noncarcinogenic health effects and summarizing other relevant toxicity information. Chemical-specific models for evaluating lead exposures and risks were also used.

For assessing carcinogenic health effects, carcinogenic slope factors (CSFs) are used to estimate the incremental lifetime risk of developing cancer corresponding to estimated exposure levels calculated in the exposure assessment. The potential for noncarcinogenic health effects is typically evaluated by comparing estimated daily intakes with reference doses (RfDs), which represent daily intakes at which no adverse effects are expected to occur during chronic

exposures. Both CSFs and RfDs are specific to the route of exposure (e.g., ingestion [oral] exposure).

As indicated in U.S. EPA (2003d) guidance, the primary source for EPA-derived toxicity values is EPA's Integrated Risk Information System (IRIS) (U.S. EPA 2009d). This computerized database contains verified toxicity values and up-to-date health risk and EPA regulatory information for many chemicals commonly detected at hazardous waste sites. The EPA-derived toxicity values used to assess carcinogenic and noncarcinogenic health effects of the CoIs are summarized in Tables 6-14 and 6-15, respectively. The values presented in these tables reflect the most recent information available in IRIS at the time of preparation of this HHRA.

Tables 6-14 and 6-15 also present additional toxicity information regarding the basis and certainty of these values. For carcinogens, this information includes EPA's weight-of-evidence classification, which indicates the adequacy of the data supporting designation of a chemical as a carcinogen. For chemicals that EPA has classified as Group A carcinogens (i.e., chemicals that have been shown to cause cancer in humans), the type of cancer associated with the chemical exposure is also listed. For noncarcinogens, the table lists the basis for the RfDs, the health endpoints associated with each RfD, and EPA assessments of the overall quality of the database supporting the RfDs.

As noted above, because of certain unique features of lead toxicity and exposure assessment, no standard EPA toxicity factors are available for lead and risk estimates for this chemical are not derived by standard methods. Instead, special models of lead exposure and biokinetics have been developed specifically for this chemical and were applied at this Site.

Supplemental toxicity information for CoIs at the Site is summarized below. Uncertainties associated with toxicity assessment are discussed in Section 6, *Uncertainty Analysis*.

6.4.1 Supplemental Toxicity Information for Antimony

Antimony, a silvery white metal, is a naturally occurring element and is ubiquitous in the environment in trace amounts. Antimony is included in alloys in the metals industry and is used for producing fireproofing chemicals, ceramic, glassware, and pigments.

Antimony is not considered a carcinogen via ingestion, but some evidence from animal studies suggests that inhalation of higher doses of antimony may induce lung tumors (ATSDR 1992). No quantitative CSF is available for antimony.

An RfD of 0.0004 mg/kg-day has been verified by EPA based on a study in which potassium antimony tartrate in water was administered orally to rats. A lowest observed adverse effects level (LOAEL) of 0.35 mg/kg-day was determined based on increased blood glucose levels and mean heart weight in males. In addition, significantly shorter life spans and altered cholesterol levels were noted in both males and females treated with potassium antimony tartrate (U.S. EPA 2009d). An uncertainty factor of 1,000 is incorporated into the RfD to account for interspecies extrapolation, the lack of an established no-observed-adverse-effect level (NOAEL), and for the protection of sensitive individuals. Confidence in the study is low because only one species and one dose level were used, a NOAEL was not determined, and gross pathology and

histopathology were not well described. Other studies on mice report similar effects, including reduced lifespan and glucose and cholesterol metabolism deficiencies; however, effects on the heart were not observed (U.S. EPA 2009d).

There are no quantitative human data concerning the absorption of antimony from the lungs, the gastrointestinal tract, or following dermal exposure (ATSDR 1992). Animal data suggest that gastrointestinal absorption of antimony tartrate and antimony trichloride ranges from 2 to 7 percent (ATSDR 1992). ICRP (1981, as cited in ATSDR 1992) recommends human reference values for gastrointestinal absorption of 10 percent for antimony tartrate and 1 percent for other forms. The absorption and toxicity of antimony found in soil at the Site may be lower than that of the soluble form used in the animal study that forms the basis for the RfD. As a result, because the RfD was used in this HHRA without adjusting the exposure estimates for potential differences in the relative bioavailability of antimony in soil, the actual risk posed by exposures to antimony in Site soil is likely to be overestimated.

6.4.2 Supplemental Toxicity Information for Arsenic

Arsenic is a naturally occurring element in the earth's crust and is usually found combined with one or more other elements such as oxygen, chlorine, and sulfur. Arsenic forms a variety of inorganic and organic compounds. Organic forms are less common in the terrestrial environmental and are less toxic to mammals. For these reasons, EPA has developed quantitative toxicity factors only for inorganic arsenic.

EPA toxicity values that are available for ingested arsenic include an RfD and a CSF. The oral RfD for arsenic of 3×10^{-4} (mg/kg-day) is based on the observation of Blackfoot disease, a peripheral vascular disease, in an epidemiological study of a Taiwanese population consuming drinking water containing naturally elevated arsenic concentrations (Tseng et al. 1968; Tseng 1977; U.S. EPA 2009d). EPA has stated that various other values within a factor of 2 or 3 of the currently recommended RfD value (i.e., 0.1 to 0.8 μg/kg-day) can also be supported by the existing data and application of the EPA RfD methodology. EPA estimates that the RfD methodology, by definition, yields a number with inherent uncertainty typically spanning perhaps an order of magnitude. As discussed below, other aspects of the epidemiological data supporting the RfD suggest that the degree to which the arsenic RfD may overestimate actual risks may be even greater.

EPA classifies arsenic as a Group A human carcinogen (U.S. EPA 2009d). Chronic arsenic ingestion at high doses has long been associated with increased incidence of skin cancers (U.S. EPA 1988). Now arsenic ingestion is also acknowledged as increasing the incidence of a variety of internal cancers as well (U.S. EPA 2009d; Chen et al. 1997); however, available data on internal cancer incidence is not adequate to support dose-response analysis (U.S. EPA 2009d).

Based on data regarding the prevalence of skin cancers in the same study population that forms the basis of the oral RfD, EPA has calculated an oral CSF for arsenic of 1.5 $(\text{mg/kg-day})^{-1}$ (U.S. EPA 2009d). The CSF was derived using the multistage Weibull model, which is a single-segment model that assumes a biological mechanism that causes the dose-response curve to

increase smoothly with dose, and to be linear at low doses (Brown et al. 1997b). This extrapolation method may overestimate risk at low concentrations if the dose-response curve is sublinear (i.e., dose increases within the low-dose range are associated with smaller increases in carcinogenic response than are similar size dose increases in the higher-dose range).

The uncertainties associated with the dose-response assessment for ingested arsenic using the Taiwanese data and with the application of these data to U.S. populations have been described by EPA in great detail (U.S. EPA 1988, 2009d; Thomas 1988). Moreover, the derivation of the CSF has undergone extensive review both within and outside EPA and continues to be the subject of evaluation (U.S. EPA 1998; Chappell et al. 1997; Brown et al. 1997a,b). A peer-reviewed expert panel assembled by EPA recently concluded that the mechanisms by which arsenic is thought to be carcinogenic are consistent with a sublinear dose-response in the low dose range (ERG 1997). The IRIS file for arsenic was recently revised to acknowledge the likelihood that the dose-response curve is sublinear (U.S. EPA 2009d). The conclusions in the IRIS file are supported by new scientific evidence strongly indicating that the dose-response relationship for arsenic is nonlinear and new studies demonstrating that the Taiwanese epidemiology studies of inorganic arsenic ingestion are inadequate to predict the dose-response for arsenic-induced cancers at doses of interest in the United States (Brown et al. 1997a,b; Guo 1997; Brown and Abernathy 1997; Brown and Chen 1995). Factors contributing to uncertainty in low-dose extrapolation of the carcinogenic effects of ingested arsenic are discussed in more detail in Section 6.6.

As noted above, arsenic exists in many organic and inorganic forms and in many matrices. Soluble forms of arsenic (arsenates and arsenites) are generally well absorbed in humans and animals, but the degree of absorption is reduced for insoluble compounds such as arsenic trisulfide or lead arsenate (ATSDR 2007a). Arsenic in soil is typically less readily absorbed (Ruby et al. 1996; Valberg et al. 1997; Roberts et al. 2002, 2007; U.S. EPA 2005). Because the study that forms the basis for EPA's oral toxicity factors for arsenic reflects ingestion and absorption of arsenic dissolved in water, relative bioavailability is another important factor influencing estimates of arsenic exposure and risk. For arsenic, such reductions in the absorption of arsenic from soils, relative to soluble forms, have been demonstrated in several animal models including rabbits, swine, dogs, and monkeys (Freeman et al. 1993, 1995; Groen et al. 1994; Ruby et al. 1996). These studies have resulted in relative absorption factors (RAFs) for arsenic in soil that range from 0.14 to 0.47; i.e., in the case of the RAF of 0.47, arsenic in soil is absorbed approximately one-half as well as soluble forms of arsenic. As a result, because the oral toxicity factors for arsenic were used in this HHRA without adjusting the exposure estimates for potential differences in the relative bioavailability of arsenic in soil, the actual risk posed by exposures to arsenic in Site soil is likely to be overestimated.

6.4.3 Supplemental Toxicity Information for Lead

Lead is ubiquitous in the environment, originating mostly from anthropogenic sources. Such sources (including historical sources) include lead from vehicle emissions, lead-containing paint, drinking water pipes, and solder in food cans. Many of these sources have been substantially reduced over the past two decades. The predominant exposure pathways of concern for lead are inhalation and ingestion. Occupational exposure is generally via inhalation

where airborne sources of lead are present. By contrast, the general population is usually exposed through ingestion of lead in a variety of sources, including food and water. The toxic effects of absorbed lead are equivalent regardless of the route of exposure (ATSDR 2007b).

Inorganic lead is classified as a Group B2 carcinogen (probable human carcinogen) based on sufficient evidence in laboratory animals. Numerous studies in rats and mice have shown statistically significant increased incidences in renal tumors with dietary and subcutaneous exposure to several soluble lead salts (U.S. EPA 2009d). Animal assays provide reproducible results in several laboratories and in various rat strains, with some evidence of multiple tumor sites. Human evidence of carcinogenicity is inadequate because available studies lack quantitative exposure data, as well as information on the possible risk contribution from cigarettes (U.S. EPA 2009d). The EPA Carcinogen Assessment Group has determined that a numerical CSF should not be developed or used for quantitative risk assessment for lead (U.S. EPA 2009d).

Meaningful oral and inhalation RfDs cannot be developed for lead because of its ubiquity in the environment from multiple sources. As a result, toxic effects of lead are typically correlated with observed or predicted blood lead levels rather than with calculated exposure levels. Similarly, lead risks typically are assessed by predicting the body burden of lead (as reflected in blood lead concentrations) that will result from assumed intake and uptake of lead from a variety of sources. Chemical-specific models have been developed to assess lead exposures and risks. As discussed in Section 6.3.4.1, *Quantifying Exposures for Lead*, such models have been used in this HHRA.

Adverse health effects resulting from chronic exposure to high levels of lead have been observed in occupationally exposed populations and children. Blood lead levels in excess of 30–40 μg/dL are associated with encephalopathy, gastrointestinal effects (colic), anemia, nephropathy, and abnormalities on electrocardiograms. Furthermore, high exposure to lead may cause spontaneous abortion in women and decreased fertility in men. Lower exposure levels in humans (i.e., blood lead concentrations less than 30 to 40 μg/dL) have been associated with potential adverse health effects including effects on heme synthesis, neurobehavioral effects, and effects on growth in children; and increased blood pressure in middle-aged men (ATSDR 2007b). A blood lead concentration of 10 μg/dL has been identified by the Centers for Disease Control (CDC) as the minimum blood lead concentration in young children (primarily children who are less than 6 years old) at which some type of follow-up is recommended (CDC 1991, 1997, 2009). Recent evaluations (e.g., as summarized by U.S. EPA 2006) indicate that the weight of evidence suggests that there is substantiation of neurocognitive effects in children in the blood lead range of 5–10 μg/dL. However, current policy from EPA targets a goal of no more than 5 percent of the population exceeding a blood lead level of 10 μg/dL (U.S. EPA 2009e). The current Occupational Safety and Health Administration (OSHA) guidelines for occupational exposures state that the blood lead concentrations in workers planning to have children should remain less than 30 μg/dL.

No target blood lead concentrations have been established for older children (e.g., the 9- to 18-year-old age range considered as part of the recreational user/trespasser scenarios). Most studies of the toxic effects associated with lead have focused on adults or young children and little information is available for individuals of intermediate ages. Because the primary focus of

evaluations of children's health risks from lead is typically the potential for adverse neurological effects, this effect is likely to remain the primary health effect of concern in this age group as well. In general, the susceptibility of individuals in this intermediate age group to neurological effects would be expected to decrease with age. This decrease would result because the nervous system in this age range is more developed and less vulnerable to the effects of lead than the rapidly developing systems of very young children. As a result, using a target blood lead concentration determined to be protective of fetuses and young children as the criterion for assessing potential health risks in older children would provide a conservative approach (U.S. EPA 2003c). This approach was applied in risk evaluations presented in Section 6.5, *Risk Characterization*.

Oral absorption of lead is generally low in humans. The extent and rate of gastrointestinal absorption are affected by fasting status and the specific lead salt's solubility in gastric acid. The extent of lead absorption is also dependent on age. For example, as part of developing a physiologically-based model for lead disposition in adult humans, O'Flaherty (1993) estimated that absorption of dietary lead ranges from 4 –11 percent, with a mean value of approximately 8–9 percent. A value of 8 percent was found to work well in the physiologically-based model to predict lead absorption from food and water. In contrast, dietary lead absorption by children through the gastrointestinal tract is generally estimated as approximately 50 percent (U.S. EPA 1994d).

The difference in the absorption of lead in soil relative to lead absorption from other media (such as food and water) is widely recognized in lead exposure models. For example, in EPA's lead exposure model for children, the IEUBK model (U.S. EPA 1994d), young children's absorption of lead from soil and dust is assumed to be 30 percent, while that from food and water is assumed to be 50 percent, yielding a relative bioavailability factor for soil relative to water of 60 percent (30 ÷ 50). A similar relative bioavailability factor for lead in soil is assumed in EPA's recommendations for evaluating lead exposures in adults (U.S. EPA 2003a).

As described in Section 4.7, *Lead Particle Analysis Results*, some Site-specific evaluations were undertaken to assess the relative bioavailability of the types of lead found at the Site. These analyses focused on materials collected from the OCA and examined the relationship between lead concentration and particle size in the selected samples. Specifically, seven of the samples collected from the OCA during the remedial investigation were screened using a 500-μm sieve to divide the samples into two size fractions, i.e., the particles greater than 500 μm and those less than 500 μm.

The 500-μm size cutoff was selected based on the potential for these size fractions to contribute to overall exposures in human and ecological receptors. Specifically, the particles in the larger size fraction should not contribute to human and ecological exposures through direct contact and ingestion or through inhalation. Human exposures through direct contact and ingestion are considered to occur only for particles that are less than 250 μm in size, because larger particles do not adhere to the hands (Duggan 1985). As a result, samples sieved to include only those particles smaller than 250 μm have been used by EPA and others in a number of evaluations of lead bioavailability (e.g., U.S. EPA 1996a, Maddaloni et al. 1996) and in exposure studies relating environmental lead concentrations to observed blood lead measurements (e.g., Butte-Silver Bow DOH and UCDEH 1992; PTI 1994). Consideration of similar factors for ecological

receptors led to selection of the 500-μm size cutoff as a reasonable criterion for determining particle sizes that might contribute to overall exposures.

The analytical results shown in Table 5-3 indicate that lead concentrations are very similar within these two size fractions, although samples with lead concentrations greater than 500 mg/kg exhibit slightly greater lead concentrations in the less-than-500-μm size fraction relative to the greater-than-500-μm size fraction. Because most of the sample mass in the seven OCA samples resides in the coarse fraction, however, most of the lead mass (i.e., 88 percent on average) is present in the greater-than-500-μm size fraction. These results suggest that although exposures to lead in the OCA sediments can be accurately characterized by analysis of bulk lead concentrations, the actual mass of lead available to human or ecological receptors is more limited than the bulk data would suggest.

6.4.4 Supplemental Toxicity Information for Carcinogenic PAH Compounds

PAH compounds are a class of organic compounds that share common structural features (two or more joined aromatic rings) and similar toxicological, physical, and chemical properties. PAHs are formed naturally during forest fires and combustion of coal, oil, gas, garbage, or other organic substances, and comprise over 85 percent of creosote and are constituents of many petroleum products. PAHs have been found in coal tar production plants, coking plants, bitumen and asphalt production plants, coal-gasification sites, smoke houses, aluminum production plants, coal tarring facilities, and municipal trash incinerators. PAHs can be found throughout the environment in the air, water, and soil (ATSDR 1995). PAHs typically do not occur alone, but rather are found in complex mixtures of many compounds. The primary sources of exposure to PAHs for most of the U.S. population are inhalation of the compounds in tobacco smoke, wood smoke, and ambient air, and consumption of PAHs in foods.

Toxicological studies on PAHs have been performed both on various PAH-containing mixtures and on individual PAH compounds. EPA has classified the following seven PAHs as probable human carcinogens (Group B2) based on sufficient data in animals: benzo[a]pyrene, benz[a]anthracene, benzo[b]fluoranthene, benzo[k]-fluoranthene, chrysene, dibenz[a,h]anthracene, and indeno[1,2,3-cd]pyrene. All of these compounds produce various tumors by several different routes of administration in animals. Benzo[a]pyrene, the most extensively studied PAH, causes skin, respiratory tract, stomach, and digestive tract tumors by oral, intratracheal, inhalation, and dermal routes of administration in rodents and monkeys. PAHs generally cause cancer at the first site of contact (i.e., oral exposure tends to result in stomach cancer and dermal exposure typically results in skin cancer). Although there are no human data that specifically link exposure to any of these PAH compounds with human cancer, all of these compounds are components of mixtures that have been associated with human cancer, such as coal tar, soots, coke oven emissions, and cigarette smoke.

Benzo[a]pyrene is the only cPAH for which oral exposure data adequate to support a dose-response assessment for carcinogenicity are available. EPA's current oral CSF for benzo[a]pyrene is 7.3 $(mg/kg/day)^{-1}$ (U.S. EPA 2009d). The slope factor is a geometric mean of four slope factors derived from two studies in experimental animals. EPA's standard default

model for carcinogenic dose-response assessment (i.e., the linear multistage model) was applied using data from a feeding study in rats (Brune et al. 1981, as cited in U.S. EPA 2009d) to derive one of the slope factor estimates. Data from a second feeding study in mice that demonstrated an excess incidence of forestomach tumors (Neal and Rigdon 1967, as cited in U.S. EPA 2009d) were analyzed using two alternative dose-response models to calculate the three additional slope factors.

EPA currently recommends use of a relative potency estimate (RPE) approach to quantitatively account for observed differences in carcinogenic potency among PAH compounds. Estimates of the relative potency of the cPAH compounds were developed based on the results of mouse studies in which benzo[a]pyrene and at least one other carcinogenic compound were investigated. The route of exposure to PAHs in these studies was skin application. EPA has issued draft provisional guidance on the relative potency approach, which recommends that the approach be used as interim guidance for risk assessment of PAH compounds (U.S. EPA 1993a,b).

This guidance specifies the following RPE values for the seven PAH compounds that EPA considers potential human carcinogens:

- Benzo[a]pyrene 1.0
- Benz[a]anthracene 0.1
- Benzo[b]fluoranthene 0.1
- Benzo[k]fluoranthene 0.01
- Chrysene 0.001
- Dibenz[a,h]anthracene 1.0
- Indeno[1,2,3-cd]pyrene 0.1

The RPE approach can be applied by adjusting the measured concentrations of individual cPAH compounds to reflect their potencies relative to benzo[a]pyrene, the reference PAH compound (i.e., by multiplying the measured PAH concentration by the RPE value for that compound). The CSF for benzo[a]pyrene can then be used with these adjusted concentrations to develop estimates of carcinogenic risk. Alternatively, the RPE values can be multiplied by the slope factor for benzo[a]pyrene to result in the following CSFs for the other six cPAHs:

- Benz[a]anthracene 0.73
- Benzo[b]fluoranthene 0.73
- Benzo[k]fluoranthene 0.073
- Chrysene 0.0073
- Dibenz[a,h]anthracene 7.3
- Indeno[1,2,3-cd]pyrene 0.73.

Currently, EPA has not determined oral RfDs for noncarcinogenic effects for the seven cPAHs of interest.

Similar to the other Site CoIs, available data suggest that the absorption of PAHs from soil is less than the absorption from the exposure medium used in the experimental study which forms the basis for the CSF for cPAHs. Specifically, the CSF for benzo[a]pyrene (which is adjusted for use with the other cPAHs as well) was derived from two rodent studies in which benzo[a]pyrene was dosed to the animals in the diet (i.e., laboratory chow). Several studies have indicated, however, that the oral absorption of benzo[a]pyrene and other PAHs from soil is considerably less than absorption from laboratory chow. For example, a study of the bioavailability of benzo[a]pyrene in male Sprague-Dawley rats indicated that the relative absorption fraction for benzo[a]pyrene (relative to absorption from chow) was 0.57 for sandy soil and 0.37 for clayey soil (Goon et al. 1991). The absorption of PAHs has also been studied in female CD1 mice using soils contaminated with manufactured gas plant residues (Weyand et al. 1996, Rozett et al. 1996). As summarized in Magee et al. (1996), data from the preceding three studies indicate that the average relative absorption fraction for PAHs from the soils tested to date is 0.29. Because the oral CSFs for cPAHs were used in this HHRA without adjusting the exposure estimates for potential differences in the relative bioavailability of PAHs in soil, these studies indicate that the actual risks posed by cPAHs in Site soil may be overestimated by approximately three-fold.

6.5 Risk Characterization

In risk characterization, a quantitative evaluation of potential health risks is performed. For lead, the blood lead concentration estimates predicted by chemical-specific models are compared with benchmark values. For other CoIs, quantitative exposure estimates and toxicity factors are combined to calculate numerical estimates of health risk. In addition, this phase of the risk assessment involves interpreting and qualifying the derived risk estimates. In this section, the results of applying EPA's recommendations regarding evaluations of lead exposures are assessed (U.S. EPA 2003a). Cancer and noncancer health risks are estimated for the Site, assuming long-term exposure to chemicals detected at the Site. For CoIs other than lead, the risk characterization methods described in EPA guidance (U.S. EPA 1989) are used to calculate upper-bound excess lifetime cancer risks for potential carcinogens and hazard indices for chemicals with noncancer health effects. These methods and the results of the risk characterization are described below.

6.5.1 Approach for Lead

To assess the potential health risks associated with lead exposures, the predicted blood lead concentrations derived using chemical-specific models are compared with blood lead concentration goals. For assessing adult lead exposures, EPA has selected protection of fetuses born to women exposed under the conditions assumed in the model as the objective (U.S. EPA 2003a). Based on this determination, the concentration goal is that the predicted 95th percentile blood lead concentration for fetuses be less than 10 μg/dL, the minimum blood lead concentration in young children at which the CDC recommends some type of follow-up (CDC

1991, 1997). Use of this goal is interpreted as meaning that there is a 95-percent probability that an exposed fetus would have a blood lead concentration lower than the target concentration of 10 μg/dL (U.S. EPA 2003). Assuming that the ratio between the fetal and maternal blood lead concentrations is 0.9, the maternal blood lead concentration corresponding to this target is 11.1 μg/dL (i.e., 10 μg/dL ÷ 0.9). This approach is more conservative than the current OSHA guidelines for occupational exposures, which state that the blood lead concentrations in workers planning to have children should remain less than 30 μg/dL. At the request of OEPA, comparisons to additional (lower) target blood lead levels have also been provided for the worker scenario, as further discussed in Section 6.6.3.

The 10-μg/dL benchmark was also used to evaluate the results obtained for the older children (ages 9–18 years) examined as part of the recreational user/trespasser scenarios and as a component of the hypothetical residential scenario in Appendix I. For the recreational user/trespasser scenarios, comparisons to additional (lower) target levels are provided in Section 6.6.3.

For hypothetical residential exposures for a young child (0–6 years of age), a target blood lead level of 10 μg/dL was also used. EPA considers a blood lead concentration of 10 μg/dL as a level of concern for young children and uses 5 percent as the regulatory limit for the probability of exceeding 10 μg/dL (U.S. EPA 1994d, 2003a). Additional discussion for this scenario is provided in Appendix I.

6.5.2 Approach for Other CoIs

For CoIs other than lead, carcinogenic and noncancer health risks were evaluated. Quantifying total excess cancer risk requires calculating risks associated with exposure to individual carcinogens and aggregating risks associated with simultaneous exposure to multiple carcinogenic chemicals. Cancer risks for a single carcinogen are calculated by multiplying the carcinogenic CDI of the chemical by its carcinogenic slope factor as follows:

$$\text{Risk} = \text{CDI} \times \text{CSF}$$

where:

Risk	=	a unitless probability (e.g., 1×10^{-6}) of an individual developing cancer over a 70-year lifetime
CDI	=	chronic daily intake averaged over 70 years (mg/kg-day)
CSF	=	carcinogenic slope factor, expressed in $(\text{mg/kg-day})^{-1}$.

A 1×10^{-6} cancer risk represents a one-in-one-million additional probability that an individual may develop cancer over a 70-year lifetime as a result of the exposure conditions evaluated. Chemical intake rates for each exposure scenario were calculated as described in Section 6.3, *Exposure Assessment*.

This linear equation is valid only at risk levels less than approximately 1×10^{-2}. No risk estimates exceeding this level were calculated for the Site.

Because cancer risks are assumed to be additive, risks associated with simultaneous exposure to more than one carcinogen in a given medium are aggregated to determine a total cancer risk for each exposure pathway (U.S. EPA 1989). Where multiple exposure pathways exist, total cancer risks for each pathway are then summed for reasonable combinations of exposure pathways to determine the total cancer risk for the population of concern.

In contrast with carcinogenic effects, potential noncancer effects are not expressed as a probability. Instead, these effects are expressed as the ratio of the estimated exposure over a specified time period to the RfD derived for a similar exposure period (e.g., CDI:chronic RfD). This ratio is termed a hazard quotient and is calculated as follows:

$$\text{Hazard Quotient} = \text{CDI}/\text{RfD}$$

where:

CDI = chronic daily intake

RfD = reference dose.

The estimated exposure and the reference toxicity factor are expressed in the same units and represent the same exposure period (i.e., chronic, subchronic, or shorter-term). Where available, toxicity factors for the noncarcinogenic effects of carcinogenic chemicals were also included. If the chemical-specific CDI exceeds the RfD (i.e., the hazard quotient is greater than 1), noncancer adverse health effects may be a concern. Exposures resulting in a hazard quotient that is less than or equal to 1 are very unlikely to result in noncancer adverse health effects.

Hazard quotients for individual chemicals are summed for each exposure pathway to determine a noncancer hazard index as follows:

$$\text{Hazard Index} = \text{CDI}_1/\text{RfD}_1 + \text{CDI}_2/\text{RfD}_2 + \ldots + \text{CDI}_i/\text{RfD}_i$$

where:

CDI_i = chronic daily intake for the ith toxicant

RfD_i = reference dose for the ith toxicant.

Where multiple exposure pathways exist, hazard indices for each exposure pathway are then summed for reasonable combinations of exposure pathways to determine a total hazard index.

Typically, where a hazard index exceeds 1, an additional analysis is considered in accordance with EPA guidance (U.S. EPA 1989). Specifically, the target organ of the effect used as the basis for the reference toxicity factor (i.e., the critical effect) is reviewed. Where this review indicates the potential for the primary contributors to the calculated total hazard indices to operate via different target organs or mechanisms of toxicity, this potential is noted and the calculated hazard indices may be modified to reflect values associated with specific target organs or effects. The uncertainty factor associated with each toxicity factor is also reviewed to determine the validity of summing hazard quotients. Because no hazard indices calculated for this Site exceeded the target value of 1, these additional evaluations were not required.

Estimates of the total cancer risk and the total hazard index associated with each of the scenarios evaluated quantitatively for the Site are presented below. Section 6.5.3 presents the results for a current onsite worker with assumed exposure to CoIs in soil, sediment, and sludge from various Site areas. Quantitative analyses of potential risks for recreational users/trespassers with exposures in the South Ditch or OCA are presented in Section 6.5.4. For each scenario, risk assessment results are presented separately for lead and for other CoIs. Section 6.5.5 summarizes the results of the risk characterization. Following EPA guidance, numerical estimates of risk are rounded to one significant figure to reflect the level of certainty associated with calculated risks. The results of the risk analyses are followed by an uncertainty analysis section (Section 6.6), which identifies the major sources of uncertainties associated with the risk analyses presented in this section.

6.5.3 Risk Characterization for Onsite Workers

6.5.3.1 Results for Lead

Table 6-16 summarizes the predicted blood lead concentrations for the onsite worker scenario. These values reflect the estimated 95th percentile blood lead concentrations predicted for fetuses of female workers exposed under the assumed conditions. Exposure concentrations were examined for each of the three Site areas of interest, i.e., the East Swale, the EFA sludge, and the South Ditch, using the arithmetic mean concentration for all data collected from the area. Additionally, depth-specific arithmetic means were calculated for each 6-in. interval up to 2 ft for each area. The incorporation of these depth-specific values, as requested by OEPA, is intended to encompass potential future scenarios that might allow deeper soils to reach the surface (e.g., septic system repair, gardening) (U.S. EPA 2003b). In addition, concentrations for specific locations of interest (e.g., the vicinity of NPDES Outfall 001 and 004 on the South Ditch) were also assessed (Table 6-16, Figure 1-2).

Assuming weekly visits by workers to the specified Site areas, along with the default soil ingestion rate, the predicted 95th percentile fetal blood lead concentrations range from 2.4–6.9 μg/dL for all three Site areas at all depth ranges, including both outfalls in the South Ditch (Table 6-16). All of these predicted concentrations are well below EPA's target blood lead concentration of 10 μg/dL. Under the assumption of more frequent exposure for hypothetical future worker scenarios, the 95th percentile blood lead levels do not exceed EPA's target blood lead concentration of 10 μg/dL for the East Swale or the South Ditch at any depth interval.

For the sludge in the EFA, using the default soil ingestion rate, all exposure frequencies greater than weekly produced a 95th percentile blood concentration above the target blood lead level of 10 μg/dL (Table 6-16). As discussed in Section 6.2.2.1, the "sludge pits" in the EFA were covered with approximately 2 ft of soil in October 1980. Thus, these materials are currently covered and the soil data from the EFA (with chemical concentrations below screening levels) accurately represent the current conditions for exposure potential in this area.

Calculations performed using the contact-intensive soil ingestion rate of 100 mg/day for concentrations found in EFA sludge resulted in 95th percentile blood lead concentrations above the target for all exposure frequencies evaluated. However, for the other Site areas evaluated

under the worker scenario (i.e., East Swale and South Ditch), calculated values exceeded the target of 10 μg/dL only when assuming daily exposure to certain areas or depths (Table 6-16).

Finally, as requested by OEPA, comparisons to additional lower target blood lead levels for the East Swale, the EFA sludge, and the South Ditch under the onsite worker scenario are provided in Section 6.6.3.

6.5.3.2 Results for CoIs Other Than Lead

Table 6-17 summarizes the total cancer risk estimates and hazard indices for the Site areas evaluated under the current onsite occupational scenario. The values shown in this table reflect the total risks obtained by summing the risk estimates derived for each of the chemicals considered in the quantitative risk analyses. The specific results generated for each chemical are shown in the more detailed tables supplied in Appendix F.

Based on EPCs derived using all Site data for a specified Site area, and assuming monthly visits to the specified Site areas, the cancer risk estimate associated with incidental ingestion of soil, sediment, and/or sludge is 2×10^{-6} for the South Ditch area, 1×10^{-6} for the East Swale area, and 2×10^{-6} for the sludge concentrations in the EFA. These risk estimates are all below OEPA's cumulative excess lifetime carcinogenic risk goal of 1×10^{-5} (OEPA 2004b).

Assuming a weekly exposure frequency, the calculated cancer risks are 8×10^{-6} for the South Ditch area, 6×10^{-6} for the East Swale area, and 9×10^{-6} for the sludge concentrations in the EFA. All of these values are less than OEPA's cumulative excess lifetime carcinogenic risk goal of 1×10^{-5} (OEPA 2004b). Even if a worker encountered all of the three Site areas of interest, the carcinogenic risk estimates would remain well within EPA's acceptable risk range of 1×10^{-6} to 1×10^{-4} established in the National Contingency Plan for Superfund sites. The total cancer risk estimated for exposures to all three areas (assuming that all three areas are visited on a monthly basis) is 6×10^{-6}, while the total estimate calculated assuming weekly visits to all three areas is 2×10^{-5}.

Using the EPC values calculated based on all Site-area data, the noncancer hazard indices associated with monthly exposures to the Site areas of interest are 0.01 for the South Ditch area, 0.02 for the East Swale area, and 0.02 for the sludge concentrations in the EFA. Assuming weekly exposures, the estimated hazard indices are 0.05 for the South Ditch area, 0.08 for the East Swale area, and 0.07 for the sludge concentrations in the EFA. All of these values are well below EPA's and OEPA's target hazard index of 1. Even if a worker were to visit all three Site areas of interest, the total hazard index would remain less than the target value. The total indices would be 0.05 (assuming monthly visits to all three areas) and 0.2 (assuming weekly visits to all three areas).

Arsenic is the only carcinogenic CoI identified for the East Swale and the EFA sludge. cPAHs were also identified as CoIs for the South Ditch; however, arsenic was the primary contributor to total carcinogenic risk estimates in this area. Arsenic and antimony were the CoIs associated with noncancer health effects identified for all three areas. In nearly all cases, arsenic was the

primary contributor to the estimated hazard indices, although arsenic and antimony were equal contributors to the estimated hazard indices in the East Swale.

The effects of applying EPCs based on various subsets (e.g., various depths or for specific subareas) of the Site data were also evaluated. The alternative EPCs that were calculated are shown in Appendix F. For the EFA sludge, no data subsets were examined. Because the sludge has been covered, all of the data for this material represent samples collected from the subsurface.

For the East Swale, the arsenic EPC based on the surface data (0–6 in.) only is greater than the EPC used in the risk calculations by a factor of 1.8. The antimony EPC based on the surface data is a factor of 1.4 greater than the EPC used in the risk calculations. For the South Ditch, the arsenic EPC based on all data from the South Ditch was less (by a factor of 1.4) than the maximum EPC for the subsets that were evaluated. For antimony in the South Ditch, the EPC used was the maximum value of 51.5 mg/kg, which was observed in the surface. Use of these values would only slightly modify the total risk values derived in the risk calculations. Using the highest EPC for each chemical for each area, the total carcinogenic risk for a worker who visited all three areas on a monthly basis would increase slightly to 7×10^{-6}, while that based on assumed weekly visits would increase slightly to 3×10^{-5}. The hazard indices for such a worker would also increase slightly to 0.06 assuming monthly visits and 0.3 assuming weekly visits to all areas of interest. Thus, the exposures and risks calculated based on these values would not be significantly greater if a different data subset were used in the calculation.

6.5.3.3 Summary of Results for Onsite Worker Scenario

For the East Swale and South Ditch, the results of the evaluations for lead for onsite workers indicate that the predicted 95th percentile blood lead concentrations were well below EPA's benchmark value of 10 μg/dL for any exposure frequency when assuming a default soil ingestion rate (Table 6-16). When assuming a contact-intensive soil ingestion rate for these two areas, only daily exposures were associated with exceedances of the target blood lead level. For the EFA sludge, the predicted 95th percentile blood lead concentrations were well below EPA's benchmark value of 10 μg/dL for weekly visits using a default soil ingestion rate, but were above the target value for all other scenarios (Table 6-16). However, the EFA sludge is currently covered with approximately 2 ft of soil, which has chemical concentrations below screening levels.

The cancer risk estimates generated for each area are well within EPA's acceptable risk range and are below OEPA's target risk value. The hazard indices are also less than EPA's and OEPA's target level of 1. Actual exposures to metals in soils would be less than the calculated values because no adjustments were made for the likely reduced gastrointestinal absorption of arsenic and antimony from soil relative to the absorption of the soluble forms of these chemicals. Because of the generally isolated locations of the affected soils, and the anticipated continuation of commercial/industrial uses of the prior industrialized portion of the property, similar low risk levels would be expected for workers in the future.

As previously noted, Section 6.6.3 provides comparisons to additional, lower target blood lead levels for the East Swale, EFA sludge, and the South Ditch under the onsite worker scenario as requested by OEPA.

6.5.4 Risk Characterization for Current and Hypothetical Future Recreational Users/Trespassers

6.5.4.1 Results for Lead

Tables 6-18 and 6-19 summarize the predicted blood lead concentrations for the recreational user/trespasser scenarios in the South Ditch and OCA, respectively. For the adults assumed to be exposed in this scenario, these concentrations were calculated to reflect the 95th percentile blood lead concentration in fetuses of mothers exposed under the assumed conditions. For the older children assumed to be exposed in this scenario, these concentrations were calculated to reflect the 95th percentile blood lead concentration in these individuals if exposed under the assumed conditions. Exposure concentrations were examined for the South Ditch and OCA (upper creek area, non-deltaic area, deltaic area) using the arithmetic mean concentration for all data collected from the area. Concentrations observed in surface samples (i.e., 0–6 in.) were also evaluated separately from concentrations observed in subsurface samples, for which EPCs were calculated for every 6-in. depth interval, up to a depth of 2 ft, per OEPA's request.

Because of the diversity of exposure assumptions included in the calculations (i.e., various exposure frequencies, soil ingestion rates, and soil depth intervals), numerous results were obtained. The discussion below provides a brief overview, with a concise summary of the complete set of results provided in Tables 6-18 and 6-19.

South Ditch—Assuming weekly visits of adults to the South Ditch, the predicted 95th percentile fetal blood lead concentrations ranged from 2.4–4.6 μg/dL. For older children, the range of 95th percentile blood lead concentrations predicted assuming weekly visits to the South Ditch was 2.6–5.1 μg/dL (Table 6-18). All of these predicted concentrations for weekly visits were well below EPA's target concentration of 10 μg/dL. Similarly, all predicted blood lead concentrations for both receptors assuming twice weekly visits to the South Ditch were below EPA's target concentration (Table 6-18).

Under the assumption of visits every other day to the South Ditch, all predicted 95th percentile blood lead concentrations were below EPA's target level for both the 25 mg/day and default (50 mg/day) soil ingestion rates. Using the contact-intensive soil ingestion rate (100 mg/day), along with the assumption of visits every other day, EPA's target level was exceeded by a few predicted 95th percentile blood lead concentrations associated with lead concentrations found in small areas near NPDES Outfalls 001 and 004 (Table 6-18).

Under the assumption of daily visits to the South Ditch, all predicted blood lead concentrations were below EPA's target level for the 25 mg/day soil ingestion rates. Using the default or contact-intensive soil ingestion rates, along with the assumption of daily visits, EPA's target blood lead level was exceeded by some predicted 95th percentile blood lead concentrations

associated with higher lead concentrations, primarily those found in small areas near NPDES Outfalls 001 and 004 (Table 6-18).

Offsite Creek Area—Assuming weekly visits of adults to the OCA, the predicted 95th percentile fetal blood lead concentrations ranged from 2.4–5.8 μg/dL. For older children who might visit this area, the range of 95th percentile blood lead concentrations predicted assuming weekly visits to the OCA was 2.7–6.5 μg/dL (Table 6-19). All of these predicted concentrations for weekly visits were well below EPA's target concentration of 10 μg/dL.

Similarly, all predicted 95th percentile blood lead concentrations for both receptors assuming twice weekly visits to the OCA were below EPA's target concentration, with the single exception occurring under the assumption of the highest soil ingestion rate (Table 6-19).

Under the assumption of visits every other day to the OCA, all 95th percentile predicted blood lead concentrations in the upper creek and non-deltaic area were below EPA's target level for both the lower and default soil ingestion rates. Using the contact-intensive soil ingestion rate, along with the assumption of visits every other day to the deltaic area of the OCA, EPA's target level was exceeded by a few predicted 95th percentile blood lead concentrations (Table 6-19).

Under the assumption of daily visits to the OCA, all predicted 95th percentile blood lead concentrations were below EPA's target level for the non-deltaic area. Using the contact-intensive soil ingestion rate, along with the assumption of daily visits to the upper creek area, EPA's target level was exceeded by some predicted 95th percentile blood lead concentrations. Using the default or contact-intensive soil ingestion rates, along with the assumption of daily visits to the deltaic area, EPA's target level was exceeded by several predicted 95th percentile blood lead concentrations (Table 6-19).

For the OCA, the EPCs and corresponding blood lead concentration predictions were higher for the surface soil/sediment concentrations than for the observed concentrations in the subsurface materials. The highest EPCs and corresponding predicted blood lead concentrations were associated with the surface concentrations in the OCA deltaic area below the railroad tracks.

Comparisons to additional, lower target blood lead levels for both the South Ditch and OCA are provided in Section 6.6.3.

6.5.4.2 Results for Cols Other Than Lead

Table 6-20 summarizes the total cancer risk estimates and hazard indices evaluated under the current recreational user/trespasser scenarios. The values shown in this table reflect the total risks obtained by summing the risk estimates derived for each of the chemicals considered in the quantitative risk analyses. An example of the specific results generated for each chemical are shown in the more detailed tables supplied in Appendix F.

Assuming monthly visits, the cancer risk estimates associated with incidental ingestion of soil and sediment from these areas are 9×10^{-7} for the South Ditch and 4×10^{-7} for the OCA. The estimates calculated when weekly visits are assumed are 4×10^{-6} for the South Ditch and 2×10^{-6} for the OCA. These values are less than OEPA's cumulative excess lifetime carcinogenic risk

goal of 1×10^{-5} (OEPA 2004b). The noncancer hazard indices associated with monthly exposures are 0.02 for the South Ditch and 0.01 for the OCA. Assuming weekly exposures, the estimated hazard indices are 0.07 for the South Ditch area and 0.04 for the OCA. All of these values are well below EPA's and OEPA's target hazard index of 1.

Arsenic is the only carcinogenic CoI identified for the OCA. cPAHs were also identified as CoIs for the South Ditch; however, arsenic was the primary contributor to total carcinogenic risk estimates in this area. Arsenic and antimony were the CoIs associated with noncancer health effects identified for these areas. Arsenic was the primary contributor to the estimated hazard indices.

The effects of applying EPCs based on various subsets (e.g., various depths or for specific subareas) of the OCA data were also evaluated. The alternative EPCs that were calculated are shown in Appendix F.

For antimony in the South Ditch, the EPC based on all data from the South Ditch was the maximum EPC for the subsets that were evaluated and the maximum concentration detected in any sample from the OCA. Thus, the exposures and risks calculated based on this value are the maximum that would be calculated for this chemical in this Site area. For arsenic in the South Ditch, the maximum EPC, based on subsurface data, was approximately 1.4 times higher than the EPC based on data from all depths. Use of the higher EPC for arsenic in the risk calculations would slightly increase the resulting cancer risk estimates to 1×10^{-6} assuming monthly visits, and 5×10^{-6} assuming weekly visits to the South Ditch. The estimated hazard indices would remain unchanged at 0.02 assuming monthly visits (because of rounding to one significant digit) and increase slightly to 0.08 assuming weekly visits to the South Ditch.

For antimony and arsenic in the OCA, the EPCs calculated based on the subsurface data were about double those used in the risk calculations. Use of these EPCs in the risk calculations would increase the resulting cancer risk estimates slightly to 8×10^{-7} assuming monthly visits and 4×10^{-6} assuming weekly visits to the OCA. The estimated hazard indices would increase slightly to 0.02 assuming monthly visits and 0.09 assuming weekly visits to the OCA. Again, these values remain within EPA's acceptable risk range, are less than OEPA's target carcinogenic risk level, and are less than both EPA's and OEPA's target hazard index.

6.5.4.3 Summary of Results for Recreational/Trespasser Exposure Scenarios

The results of the evaluations for lead (Tables 6-18 and 6-19) for recreational users/trespassers to the South Ditch and OCA indicate that the predicted 95th percentile blood lead concentrations were well below EPA's benchmark value of 10 μg/dL assuming weekly visits (for any soil ingestion rate) or when using a lower soil ingestion rate (for any exposure frequency). Some combinations of higher soil ingestion rates and/or more frequent exposures yielded predicted 95th percentile blood lead levels above the target level (Tables 6-18 and 6-19). Additional comparisons to lower target blood lead levels are provided in Section 6.6.3. The cancer risk estimates generated are well within EPA's acceptable risk range of 1×10^{-6} to 1×10^{-4} established in the National Contingency Plan for Superfund sites and are less than OEPA's target carcinogenic risk level (Table 6-20). The hazard indices are less than both EPA's and

OEPA's target level of 1. The likely reduced bioavailability of arsenic and antimony from soil would also reduce the calculated exposures and risk estimates. Because this assessment of the South Ditch and OCA evaluated exposure frequencies up to daily and contact-intensive soil ingestion rates, patterns of use in the future are not expected to result in any greater exposures than those evaluated in the HHRA.

6.5.5 Risk Characterization for Hypothetical Future Offsite Residents

A hypothetical future land use scenario that assumes residential development in the OCA has also been evaluated at the request of OEPA. Details of this evaluation are presented in Appendix I. Included in this evaluation are modeled blood lead levels for young children in a residential setting in the non-deltaic area, as well as an evaluation of exposures that might be associated with an older child visiting the deltaic or upper creek areas of the OCA. It was assumed that the deltaic and upper creek areas could not support residential development and, in 2007, the current property owner executed a DUR to limit the future use of the contaminated areas of the OCA (GE 2009). Under the assumption of residential development in this area, the modeled blood lead levels for young children were 2.06–2.13 μg/dL, well below the regulatory benchmark of 10 μg/dL. Under the assumption of a residential older child visiting the deltaic or upper creek areas, modeled blood lead levels did not exceed regulatory benchmarks assuming weekly visits (for any soil ingestion rate) or when using a lower soil ingestion rate (for any exposure frequency). Some combinations of higher soil ingestion rates and/or more frequent exposures yielded predicted blood leads above the target level. The matrix approach, which combined varying exposure frequencies and ingestion rates, was used in order to provide OEPA with results from a full range of potential exposure scenarios. The full set of results are summarized in Table I-4. As requested by OEPA, additional comparisons to lower target blood lead levels are provided in Section 6.6.3.

6.5.6 Summary of Risk Characterization

In assessing the potential effects of lead, this assessment found that all predicted 95th percentile blood lead concentrations calculated assuming a weekly exposure frequency were less than EPA's benchmark of 10 μg/dL for all receptors in all areas, with the single exception of a worker's exposure to EFA sludge using a contact-intensive soil ingestion rate (Tables 6-16, 6-18, 6-19, and I-4). For the onsite worker, all lead concentrations at any depth in the East Swale and South Ditch were associated with predicted 95th percentile blood lead concentrations below the EPA target when using the default soil ingestion rate (Table 6-16). When assuming the higher, contact-intensive soil ingestion rate, concentrations in the East Swale and the South Ditch resulted in exceedances of the target blood lead level for the worker only when assuming daily exposures. Contacting EFA sludge at higher exposure frequencies (i.e., more than weekly) or when assuming the contact-intensive soil ingestion rate also yielded predicted blood lead levels that exceeded the target blood lead level (Table 6-16); however, as noted previously, the EFA sludge is covered with 2 ft of soil, which has chemical concentrations below screening levels. Under the hypothetical assumption of residential development within the OCA presented in Appendix I, the modeled blood lead levels for young children were 2.06–2.13 μg/dL, well below the regulatory benchmark of 10 μg/dL (Table I-3). Under the assumption of a residential older child visiting the deltaic or upper creek areas from a home within the OCA, modeled 95th

percentile blood lead levels did not exceed the regulatory benchmark of 10 μg/dL assuming weekly visits (for any soil ingestion rate) or when using a lower soil ingestion rate (for any exposure frequency) (Table I-4). Some combinations of higher soil ingestion rates and/or more frequent exposures to the OCA yielded predicted 95th percentile blood leads above the target level (Table I-4). Similar results were obtained for the recreational user/trespasser scenarios in the South Ditch and OCA, which yielded predicted 95th percentile blood lead concentrations well below EPA's benchmark value assuming weekly visits (for any soil ingestion rate) or when using a lower soil ingestion rate (for any exposure frequency) (Tables 6-18 and 6-19). Some combinations of higher soil ingestion rates and/or more frequent exposures to the South Ditch or OCA yielded predicted 95th percentile blood lead levels above the target level for the recreational user/trespasser scenarios (Tables 6-18 and 6-19).

The risk estimates calculated for each area for carcinogenic CoIs at the Site are well within EPA's acceptable risk range and are less than OEPA's target level of 1×10^{-5} (Table 6-17). No carcinogenic risk estimates calculated for the Site exceed 1×10^{-4}. Arsenic was the only carcinogenic CoI identified for most of the areas of interest evaluated in the quantitative risk calculations. In the one case where other carcinogenic CoIs were identified for a Site area, arsenic remained the primary contributor to the estimated cancer risks. The total cancer risk estimated for the onsite worker scenario assuming contact with all three Site areas of interest is 6×10^{-6} if monthly contact is assumed and 2×10^{-5} if weekly contact is assumed (Table 6-17). Risks estimated for the recreational/trespasser scenarios in the South Ditch and OCA were approximately 2- to 3-fold less (Table 6-20).

In assessing the potential noncancer health effects associated with non-lead CoIs at the Site, the hazard indices calculated for the recreational/trespasser exposure scenarios are 1–2 orders of magnitude less than EPA's and OEPA's target hazard index of 1 (Table 6-20). The hazard indices calculated for the onsite occupational scenario are also 1–2 orders of magnitude less than EPA's and OEPA's target level for each of the three Site areas individually, and a factor of 5–20 less than the target level if it is assumed that an individual worker contacts all three areas (Table 6-17). The primary contributor to noncancer hazard indices is arsenic.

6.6 Uncertainty Analysis

Because risk characterization serves as a bridge between risk assessment and risk management, it is important that major assumptions, scientific judgments, and estimates of uncertainties be described in the assessment. The certainty of the calculated risk estimates depends on the uncertainties inherent in each step of the risk assessment process. In addition, aspects of the risk characterization process itself introduce uncertainties, including those associated with adding risks or hazard quotients for multiple chemicals. Many uncertainties are generic to the risk assessment process, while others are site-specific. The potential influence of uncertainties on the risk analyses performed for the Site is qualitatively evaluated below for each of the four risk assessment steps.

The overall goal of the uncertainty assessment is to put the risk estimates into context. To accomplish this goal, factors that may tend to over- or underestimate risks are identified and the relative magnitude of uncertainty for each factor is evaluated so that the level of confidence

associated with the risk estimates is clear. The approach taken in discussing uncertainties in the risk assessment is consistent with EPA guidance encouraging better characterization of the uncertainties inherent in risk calculations and the effects of these uncertainties, as well as variability in risk input parameters, on risk analysis results (U.S. EPA 1994a, 1989).

6.6.1 Uncertainties Associated with Data Evaluation

Uncertainties associated with identifying CoIs include:

- Potential risks associated with chemicals intentionally excluded from the risk assessment
- Potential risks associated with chemicals unintentionally excluded from the risk assessment.

As presented in Section 6.2.2, *Identifying Chemicals and Areas of Interest*, all chemicals detected in soil, sediment, groundwater, and surface water at and in the vicinity of the Site were examined as PCoIs for the HHRA. Chemicals were excluded from consideration as CoIs only where the measured concentrations were less than standard risk-based concentrations, indicating negligible concern for human health risk or where the measured concentrations were comparable to typical background concentrations. Thus, exclusion of these chemicals from the quantitative risk analyses is unlikely to significantly underestimate risks associated with the Site.

Potential risks associated with chemicals unintentionally excluded from the risk analyses constitute another source of uncertainty. This source of uncertainty is unlikely to have any significant effects on the risk calculations performed for this Site. The evaluations included a comprehensive review of operations and activities to identify chemicals that could be present at the Site. Based on chemical data collected during the Site investigations and risk-based screening of these data, the primary contributors to Site-related risks were identified. These actions minimize the potential for failing to account for any significant contributors to health risks or for significantly underestimating actual total health risks associated with the Site.

6.6.2 Uncertainties Associated with Exposure Assessment

The greatest number of Site-specific uncertainties are associated with the exposure assessment. The most significant uncertainties associated with this step that may influence the results of the risk assessment include:

- Assumptions used in estimating EPCs
- Assumptions used to estimate intake variables
- Accuracy in characterizing current Site conditions and predicting future Site conditions.
- Risks associated with potential exposure pathways excluded from the risk assessment.

Specific issues for the exposure analyses performed at this Site are discussed below.

6.6.2.1 Uncertainties in Exposure Point Concentrations

As discussed in Section 6.3.4, *Quantifying Exposure*, EPCs were developed based on a comprehensive review of all data available for the Site areas of interest. Various subsets of the data (e.g., data for surface or subsurface samples or data from specific exposure points of interest) were also considered. Summary statistics for use as EPCs were developed based on the standard guidance and exposure models for each type of CoI. For lead, arithmetic mean concentrations were used as EPCs. In a prior draft of this remedial investigation, geometric mean concentrations were selected as the most appropriate estimate of the central tendency. However, at the request of OEPA and in conformance with the EPA model, arithmetic averages were used as the EPCs for lead. Arithmetic means, when compared to geometric means, are more significantly affected by a small number of high values. Thus, given the data set for this Site, which focuses on characterizing sources rather than providing data from random locations across the Site, use of an arithmetic mean EPC is likely to overestimate actual exposures that might occur at this Site. For CoIs other than lead, the 95-percent UCL of the mean or the maximum detected concentration was used to represent the EPCs. All EPCs were calculated based on reported chemical concentrations without any adjustments for background concentrations. Because natural background concentrations may contribute to observed concentrations of some of the identified CoIs (e.g., arsenic and cPAHs), this approach overestimates the actual risks associated with chemicals present due to Site-related activities.

6.6.2.2 Uncertainties in Exposure Parameter Assumptions

Other uncertainties associated with the exposure assessment include the assumptions used to estimate intake variables for individual exposure pathways and the likelihood of exposure pathways and land uses actually occurring. As discussed in Section 6.3.4, *Quantifying Exposure*, the exposure variables used in this risk assessment were selected to represent a reasonable maximum exposure case. Exposure and risk estimates were developed for the populations with the highest potential for exposure at the Site (e.g., current onsite workers and recreational users/trespassers) using conservative exposure parameters and combinations of exposures. Where available, EPA default assumptions for relevant exposure parameters were used to quantify exposures. Some of these default parameters are based on limited empirical data, e.g., regarding rates of incidental soil ingestion in adults and older children. These values are designed to provide conservative estimates of risk. Where exposure parameters were selected based on best professional judgment, such values were selected to provide plausible, but conservative estimates of exposures that are unlikely to underestimate actual exposures.

In a number of cases, conservative exposure assumptions were applied in calculating exposures and risks. For example, except for lead, intake estimates were not adjusted for the bioavailability of CoIs in soil, sediment, or sludge relative to the bioavailability of the CoI in the exposure medium used in the toxicity study that forms the basis for the quantitative toxicity factor. For both antimony and arsenic, the toxicity factors used in these analyses are based on studies where a soluble form of the chemical was ingested in drinking water. For cPAHs, the toxicity factors used in this HHRA were based on studies where PAHs were administered in the

diet. As discussed in Section 6.4, *Toxicity Assessment*, the absorption of these chemicals from soil is likely to be lower than absorption of a dissolved form from water or from the diet. For example, studies for arsenic have reported relative bioavailability factors for arsenic in soil that range from 0.14–0.47. These values indicate that the exposure and consequent risk associated with ingestion of arsenic in soil (or other soil-like media) could be a factor of 2–7 less than those presented in the HHRA (i.e., 1÷0.47=2; 1÷0.14=7).

The assumptions used for exposure frequency are also conservative in light of the actual patterns of exposure and use at the Site. Even the minimum exposure frequency evaluated in the risk analyses (i.e., one visit to a specific Site area of interest per month) is greater than actual exposure frequencies for any individual Thomson employee. As a conservative element, an exposure frequency of 1 visit per week was also evaluated. Additionally, due to the pharmacokinetics upon which the ALM is based, the minimum exposure frequency that could be evaluated for exposure of an adult or older child to lead is one visit per week. At the request of OEPA, to provide a range of exposure scenarios, the ALM was also used to evaluate exposure frequencies of twice weekly, every other day, and daily. The limitations on access to the OCA, particularly the areas with the most elevated lead concentrations, also make these exposure frequencies conservative for any individual recreational user/trespasser of this area. Under the hypothetical residential exposure analysis (Appendix I) for the OCA, the soils are expected to be frozen or snow-covered for a portion of the year, further limiting exposures not factored in the risk estimates. Because the soils are expected to be frozen or snow-covered for a portion of the year, exposures during these periods would decrease such that blood lead levels would be lower than the modeled daily estimates. Thus, the calculated exposures and risks are likely to significantly overestimate the exposures and risks for the onsite worker and hypothetical residential scenarios, and likely represent a conservative RME case for the recreational user/trespasser scenario.

Uncertainty also exists in some of the assumptions applied in the lead model. Little information is available to determine the appropriate GSD for use in the lead models, particularly the adult model or for specific populations (i.e., geographic- or ethnic-specific). In this analysis, a GSD of 1.8, based on nationwide data was used, as currently recommended by U.S. EPA (2009b).

As noted in Section 6.4.3, *Supplemental Toxicity Information for Lead*, the default assumption for the absorption fraction also reflects a conservative value. The default value (0.12) was derived by assuming that, on average, approximately 20 percent of soluble forms of lead are absorbed following ingestion under a variety of conditions of fasting or recent food intake. Lead in soil is then assumed be absorbed 60 percent as well as soluble lead. Based on model validation efforts for a physiologically-based model for adult lead uptake, other research has indicated that the mean absorption value for dietary lead sources may be closer to 8 percent (O'Flaherty 1993). Because lead in soil would be expected to be less well absorbed than dietary lead, these data suggest that the typical absorption fraction for lead in soil may be less than 8 percent (i.e., an absorption fraction of 0.08) and less than EPA's default assumption (0.12). Preliminary results from studies of lead bioavailability from soil using adult volunteers also suggest lower absorption than indicated by EPA's default estimate (Maddaloni et al. 1998). Taking into account the likely relative timing of incidental soil ingestion and consumption of meals, these studies resulted in mean estimates of absorption fraction that ranged from 3 to

14 percent depending on the number of meals assumed to be consumed per day and the assumed soil ingestion pattern.

A final source of uncertainty in the exposure analyses is the use of the ALM under some of the exposure circumstances and for some of the assumed exposed populations at the Site. The model is most appropriately applied to evaluate sites under an occupational scenario where it is assumed that adult workers have ongoing, frequent exposures to a relatively constant level of lead in soil. Such conditions would allow lead intake and resulting blood lead concentrations to reach a steady-state condition. Where exposures are infrequent (as at this Site) or occur for only a short period of time, the degree of uncertainty in the model predictions increases.

Similar increases in uncertainty arise by using the model for a pre-adult population (i.e., the 9–18 year age group examined as part of the recreational user/trespasser scenarios). In this case, because many of the model assumptions reflect data for adult populations, the use of this model to predict blood lead concentrations in a younger population has a greater degree of uncertainty. To better adapt this model to a younger population, some parameter estimates might need to be increased, while others might need to be decreased. As discussed below, however, it appears that most of the default parameter estimates applied in the model are likely to result in health-protective model predictions for older children. Because of this uncertainty, a matrix approach, incorporating varying exposure frequencies combined with multiple soil ingestion rates, was used, as requested by OEPA.

Among the values that might need to be increased is the absorption fraction for lead from soil. Gastrointestinal absorption of lead is less in adults than in young children. The absorption in older children is likely to be intermediate between values for these other two age groups. Thus, in the older children age group, the value for the assumed absorption of lead from soil might lie between 0.12 (the value assumed in EPA's ALM) and 0.3 (the value assumed in EPA's model for young children).

The assumed baseline blood lead concentration might also require modification. Because of ongoing decreases in lead exposure sources, the typical lead exposures of the U.S. population have been decreasing. This decrease in exposure has been reflected in decreases in observed blood lead concentrations over time (Pirkle et al. 1994) and in lower blood lead concentrations in individuals of younger age groups (Brody et al. 1994). For example, using NHANES data from 1988–1994, the reported mean baseline blood lead concentration for all women is 1.80 μg/dL for the oldest age category summarized (ages 36–45), decreases to 1.55 μg/dL for ages 26–35, and decreases again to 1.23 μg/dL for ages 17–25 (U.S. EPA 2002a). Indeed, EPA's air quality criteria document noted that over the last 25 years, declines of more than 90 percent in the mean blood lead levels within all age and gender subgroups of the U.S. population have been measured (U.S. EPA 2006). Additionally, EPA recently updated their analysis of NHANES data, and now recommends using a baseline blood lead (geometric mean) of 1.0 μg/dL in the ALM, which is substantially lower than recommendations based on earlier data (U.S. EPA 2009b). Because even the most recent EPA recommendations incorporate NHANES data collected 5–10 years ago, the current blood lead concentrations may be even lower. Thus, the use of the mean baseline blood lead concentration of 1.0 μg/dL is likely to be health-protective.

Based on inspection of the results from EPA's IEUBK model for young children (U.S. EPA 2009a), use of the default biokinetic slope factor for adults (0.4 μg/dL per μg/day of lead uptake) is likely to be health-protective when applied to older children and may yield overestimates of predicted blood lead concentrations. When the IEUBK model is run, the resulting blood lead concentrations (in μg/dL) and total lead uptakes (in μg/day) for each of seven age categories examined in the model are provided as model output. By dividing the age-specific blood lead concentration by the total lead uptake, the BKSFs inherent in the IEUBK model can be estimated. These values are summarized in Table 6-21. The BKSF ranges from 0.55 for the youngest age group (6–12 months) to 0.29 for the oldest age group (6–7 years). The average for all age groups is 0.38. Based on these model results, the BKSF for the age group examined in the recreational/trespasser scenario would be expected to be comparable to or lower than that for the oldest age group included in EPA's IEUBK model (0.29). Therefore, using EPA's default value for the adult model of 0.4 appears to be a conservative element for evaluating potential exposures in older children.

Although there are a number of uncertainties associated with use of the ALM for the exposure circumstances and populations examined at the Site, many of the uncertainties discussed above suggest that the model is likely to provide health-protective estimates of potential lead exposures.

6.6.2.3 Uncertainties in Exposure Pathways

Exposure pathways excluded from the risk assessment represent potential sources of uncertainty. As described in Section 6.3.2, *Identifying Exposure Pathways*, a comprehensive listing of exposure pathways was considered for inclusion in the HHRA. Pathways were excluded from quantitative risk analyses only if screening evaluations indicated that they were unlikely to contribute significantly to total risks. These conclusions were based on Site-specific chemical and other data and comparisons with regulatory standards and standard risk-based screening levels. Thus, omission of these pathways is unlikely to result in any significant underestimate of Site risks.

Dermal exposure was excluded from the risk analyses for metals because this pathway is typically a negligible contributor to total exposures and risks for metals. In particular, neither of EPA's recommended exposure models for lead (the primary CoI identified at the Site) include this pathway because of its negligible influence on lead exposures. Certain cPAH compounds were the only organic chemicals identified as CoIs at the Site. Because of the limited extent of the area affected by cPAH compounds and the infrequent contact with this area of the Site, excluding a quantitative assessment of dermal contact is unlikely to affect the results of the risk assessment.

Exposures via groundwater pathways were also excluded from the quantitative risk analyses based on the screening evaluations. As described in Section 6.3.2, *Identifying Exposure Pathways*, exposures to Site-related CoIs in groundwater are unlikely because of the nature of the different aquifers underlying the Site. As a result, exclusion of this exposure pathway from the quantitative risk evaluations will not result in an underestimate of actual Site risks. In addition, measured concentrations of CoIs in groundwater samples from the EFA were less than

federal drinking water standards. Thus, even if the groundwater were to be consumed as a drinking water supply, the measured concentrations of CoIs do not exceed regulatory standards.

The screening evaluations also supported exclusion of surface water exposure pathways from the quantitative risk evaluations. Contacts with Site-related CoIs in surface water are likely to be infrequent and of limited extent and duration. In addition, measured concentrations of CoIs were less than federal drinking water standards. As a result, the more intermittent and incidental types of exposures to surface water that are likely to occur in the Site areas of interest would be well within the exposure levels allowed by the federal drinking water standards. Exclusion of exposure pathways associated with this medium from the risk analyses is unlikely to significantly underestimate actual Site exposures and risks.

6.6.3 Uncertainties Associated with Toxicity Assessment

Most of the uncertainties associated with the toxicity assessment are generic to the risk assessment process and include uncertainties related to:

- The quality of the studies used as the bases for toxicity values
- Potential differences in toxicity and absorption efficiency between humans and laboratory animals
- The applicability of studies conducted in experimental animals dosed at high levels to human exposures at lower concentrations
- The validity of the critical underlying assumption in the dose-response model for carcinogens that there is no threshold for carcinogenesis (i.e., there is no dose of a carcinogen that is not associated with a risk of cancer)
- Calculation of lifetime excess cancer risks on the basis of less-than-lifetime exposures
- Potential synergistic or antagonistic interactions of chemicals to which the same individual may be exposed.

Because of the above uncertainties, methods for developing toxicity values for use in risk assessment are designed to be conservative. For example, cancer risk estimates are likely to overestimate actual cancer risks because EPA-derived slope factors are frequently based on upper-bound risk estimates derived from carcinogenesis dose-response modeling. As a result, risk levels are unlikely to be underestimated and may be substantially overestimated by using these values. The use of uncertainty factors in the derivation of RfDs and other noncancer toxicity factors (ranging from 3 to 1,000 for chemicals evaluated at this Site) is also devised to be health-protective, accounting for limitations in available data and extrapolation from experimental conditions to exposure conditions of concern. As a result, risk estimates derived using EPA-derived toxicity factors are likely to over- rather than underestimate risk.

Chemical-specific uncertainties associated with the toxicity assessment presented in Section 6.4 include deficiencies in the toxicological databases for some Site CoIs. As described in Section 6.4, substantial uncertainties exist in the toxicity data available for several of the chemicals that are primary contributors to Site risks. For example, EPA has acknowledged the uncertainties in the toxicity assessment information for arsenic (U.S. EPA 1988, 2009d; Thomas 1988). Available data for arsenic suggest that EPA's oral toxicity factors, particularly the CSF for this chemical, may substantially overestimate actual risks associated with arsenic ingestion at low doses. Most prominent among these uncertainties are questions regarding the validity of the conclusions drawn from the Taiwanese data set that forms the basis for the oral CSF and RfD and the degree to which these conclusions are applicable to exposures experienced by U.S. populations (U.S. EPA 1998; Brown et al. 1997a,b; Chappell et al. 1997). As noted in Section 6.4.2, *Supplemental Toxicity Information for Arsenic,* new scientific evidence strongly indicates that the dose-response relationship for arsenic is nonlinear. As new studies have demonstrated that the Taiwanese epidemiology studies of inorganic arsenic ingestion are inadequate to predict the dose-response for arsenic-induced cancers at doses of interest in the United States (Brown et al. 1997a,b; Guo 1997; Brown and Abernathy 1997; Brown and Chen 1995), EPA has conducted more detailed assessments of the mode of action of arsenic to determine whether a linear or nonlinear model should be used in the dose-response assessment. EPA convened a workshop with a panel of seven experts in May 1997, requesting their opinions on "whether the body of available data regarding arsenic's mode of action is sufficient to support adoption of one response model (i.e., linear versus nonlinear) over the other in extrapolating from the relatively high levels of arsenic exposure in the Taiwanese population to the lower exposure levels the Agency will be addressing in the new drinking water standard [for arsenic]" (ERG 1997). As presented in the report on the workshop (ERG 1997), the expert panel concluded that inorganic arsenic is not a direct genotoxicant. They further concluded that there are several modes of action by which arsenic is likely to exert indirect genotoxicity. Although they could not conclusively identify the mode of action for arsenic, they did determine that all of the most likely modes of action would be associated with a nonlinear dose-response relationship.

In 1996, EPA issued draft cancer risk assessment guidelines that call for an assessment of the mode of action of a carcinogen in determining whether a linear or nonlinear dose-response model should be used (U.S. EPA 1996c). Although these guidelines are being revised in response to comments from the EPA science advisory board and the public, EPA has begun revising dose-response assessments for chemicals that are believed to act by nonlinear mechanisms (e.g., 63 Fed. Reg. 15674). On April 10, 1998, EPA revised the dose-response documentation presented in the IRIS file for ingested inorganic arsenic to include the conclusions of the expert panel, noting specifically the panel's finding that "the dose-response for arsenic at low doses would likely be truly nonlinear (i.e., with decreasing slope as the dose decreased). However, at very low doses, such a curve might be linear but with a very shallow slope, probably indistinguishable from a threshold" (U.S. EPA 2009d).

EPA has not yet revised the oral CSF for arsenic to reflect the nonlinear dose-response; however, EPA's risk assessment guidance allows substantial flexibility in the application of the CSF by providing a range of target risks that EPA may use to select cleanup levels at specific sites. The EPA IRIS database specifically cautions that these weaknesses and uncertainties in the arsenic risk assessment should be taken into account (U.S. EPA 2009d). As a result, site

managers often apply higher target risks and cleanup levels when they believe arsenic exposures at a site are low enough that no public health hazard exists.

Uncertainties in the magnitude of dietary arsenic sources have also been identified as a critical factor influencing both the calculation and interpretation of arsenic risk estimates (U.S. EPA 1998). In particular, arsenic intake from yams and rice (the primary dietary arsenic sources in the Taiwanese study population) was not completely accounted for in EPA's derivation of the CSF or RfD (Schoof et al. 1998). By underestimating the contributions of these sources of arsenic intake to total arsenic exposures, EPA's current CSF reflects an overestimate of the potential potency of the arsenic ingested in the Taiwanese drinking water supplies (Brown et al. 1997a,b). In addition, background exposures of U.S. populations to arsenic from such sources as food and water can be significant and can comprise a significant proportion of an individual's total arsenic exposure (U.S. EPA 1998).

For lead, uncertainty exists regarding the appropriate target blood lead concentration for older children (i.e., the age range of 9–18 years included in the recreational user/trespasser scenario). The target level used in the risk analysis (i.e., 95th percentile blood lead levels below 10 μg/dL) is designed to protect fetuses and young children (i.e., children less than 6 years old) from adverse health effects associated with lead exposure, particularly neurological effects. Because older children are at a less vulnerable developmental stage and are less susceptible to the adverse neurological effects of lead, use of a target blood lead concentration determined to be protective for fetuses and younger children as the criterion for assessing potential health risks in older children is likely to overestimate and unlikely to underestimate potential health risks in this population (U.S. EPA 2003c).

Additionally, there is uncertainty about the threshold for lead toxicity. A recent weight-of-evidence evaluation of the existing scientific literature on the health effects of lead exposure by EPA (provided in the air quality criteria document for lead [U.S. EPA 2006]) suggests that there is substantiation of neurocognitive effects in the blood lead range of 5–10 μg/dL, "and possibly somewhat lower." At this point, EPA's current policy has not changed, and still targets a 95th percentile blood lead level below 10 μg/dL (Beringer 2009, pers. comm.). However, there are indications of possible future revisions to this policy (U.S. EPA 2009e). To address this uncertainty, the predicted blood lead concentrations for the onsite worker and recreational user/trespasser scenarios were compared to lower benchmark levels of 5, 6, 7, 8, and 9 μg/dL (Tables 6-22 through 6-24).

For the onsite worker, weekly exposure to lead concentrations in the East Swale and South Ditch result in only a single exceedance of the lowest benchmark of 5 μg/dL when using the contact-intensive soil ingestion rate (Table 6-22). For exposure frequencies greater than weekly for an onsite worker, particularly when combined with a contact-intensive soil ingestion rate, exceedances of some of the benchmark blood lead levels were seen for concentrations in the East Swale and South Ditch (Table 6-22). For exposures of an onsite worker to the EFA sludge, weekly exposures using the default soil ingestion rate did not exceed the benchmarks of 7, 8, or 9 μg/dL; however, all other scenarios result in exceedances of all five of these lower benchmarks.

For the recreational user/trespasser, the results of the additional comparisons requested by OEPA indicate that under the assumption of weekly exposures, there are no exceedances of benchmark blood lead levels of 7, 8, or 9 μg/dL in either the South Ditch (Table 6-23) or the OCA (Table 6-24), and exceedances of lower blood lead targets (5 or 6 μg/dL) occur only when incorporating the contact-intensive soil ingestion rate. Further, there are no exceedances of any benchmark under any scenario for the non-deltaic area of the OCA. However, the use of exposure frequencies greater than weekly for the South Ditch and other areas of the OCA, resulted in exceedances of some of these lower blood lead benchmarks (Tables 6-23 and 6-24).

6.6.4 Uncertainties Associated with Risk Characterization

Aspects of the risk assessment model that introduce uncertainties include:

- The validity of adding risks or hazard quotients for multiple chemicals
- The validity of adding risks or hazard indices across exposure pathways.

At this Site, because many PCoIs and potential exposure pathways were determined to contribute negligibly to total exposures and risks, total Site risks and hazard indices reflect the summation of results from only a few chemicals and do not reflect summation across pathways. As a result, little uncertainty was added to the risk assessment results due to these factors.

In addition, risk assessment methods are designed to be highly conservative to address the uncertainties associated with each step in the risk assessment process. Thus, "true" Site risks are very unlikely to be greater than (and may be significantly less than) risks estimated using standard risk assessment methods. In this risk assessment, key factors that are likely to overestimate rather than underestimate Site risks include:

- Use of maximum or near-maximum Site concentrations as EPCs for certain risk estimates
- Use of conservative intake assumptions to estimate reasonable maximum exposures
- Use of toxicity values that incorporate uncertainty factors or other techniques devised to be health protective
- Use of conservative screening techniques.

As a result, actual Site risks are unlikely to be underestimated, and are likely to be overestimated, by the procedures applied in this risk assessment.

7 Ecological Risk Assessment

A Phase I ecological assessment was completed to evaluate habitat quality and potential ecological receptors at the Site and the OCA, and to determine whether a Phase II ecological assessment (baseline ecological assessment) is warranted. This assessment was based on a qualitative evaluation of habitat conditions, quantitative sampling of biota, evaluation of potential ecological receptors in onsite and offsite areas of interest, identification of PCoIs, assessment of the bioavailability of the PCoIs, sediment toxicity bioassays and evaluation of potential exposure pathways.

Initial Site surveys for the Phase I ecological assessment were completed in 1992, after which a preliminary ecological assessment document was submitted as an appendix to the RI/FS work plan (PTI 1995). During August and October 1997, the Site and offsite areas of interest to the remedial investigation were revisited. The objective of the 1997 Site surveys was to reexamine more current field conditions and review the conclusions of the preliminary assessment in light of these conditions and the additional data collected during the remedial investigation. Finally, in 2007, supplemental sediment sampling was performed within the OCA as part of sediment bioassay investigations. Those investigations involved the collection of surficial sediment samples (0- to 6-in.) from 28 locations previously sampled during the remedial investigation in an attempt to collect samples representative of the following concentration ranges for use in the sediment bioassays: 40 to 400 ppm, 400 to 800 ppm, and greater than 800 ppm. Based on these analyses, a final Phase I ecological assessment was completed and is provided as Appendix G. The key conclusions regarding the Phase I ecological assessment are summarized below.

The conclusion of the Phase I ecological assessment is that lead is the only CoI, and the EFA and OCA are the only habitats where potential exposure of ecological receptors to Site-related chemicals may occur. Factors influencing the evaluation of potential risk to receptors include: quality of habitat, size of the potentially affected habitat relative to the foraging area of a receptor, proportion of a receptor population affected, potential for complete exposure pathways to exist, toxicity of lead to the receptor, bioavailability of lead, and quantitative food web exposure models for the most highly exposed receptors. Based on the following factors, it was concluded that potential ecological risks associated with lead from the Site on the EFA and OCA are not significant, and do not warrant a Phase II ecological assessment:

- The quality and extent of habitat at the EFA is limited. The EFA is a small (about 5 acres), idle, mid-successional grassland habitat. While the area may support some receptors, it is unlikely to support any significant populations of receptors of interest.
- Food webs in the EFA are not expected to be significantly affected by lead concentrations in surface soils. Small wildlife receptors that potentially occur on the EFA (i.e., northern short-tailed shrew, white-footed mouse, dark-eyed junco, and America robin) are expected to be primarily granivorous, frugivorous, and insectivorous. Plant parts such as seeds and fruits are not expected to readily take up lead. For insectivorous species, foraging will

typically be on aboveground, mobile insects; foraging on subsurface invertebrates (e.g., earthworms) is expected to be uncommon except for the northern short-tailed shrew.

- The offsite creek is considered poor-quality habitat. The terrestrial habitat of the OCA is typical of transitional cover in mixed development properties. OEPA has designated the offsite creek (i.e., an extension of the South Ditch referred to as the RCA Tributary) as a "limited resource water" (OAC 3745-1-09). OEPA has determined that the aquatic habitat is of poor quality and conditions are expected to remain poor because regular channel maintenance (e.g., periodic dredging) is needed to sustain drainage of the adjacent agricultural field. This evaluation by the State is supported by observations during the 1997 Site reconnaissance that found poor habitat for receptor populations in the OCA. The OCA is subjected to additional chemical stressors from sources unrelated to the Site, including storm water runoff from commercial and residential areas, agricultural runoff, and sewage treatment plant effluent.

- The entire OCA is less than 12 acres and areas with substantially elevated lead concentrations in the deltaic area and affected channel portion of the upper creek are less than 20 percent (approximately 2 acres) of this area. Thus, the frequency and duration of the exposure of receptors to lead will be minimal on the OCA.

- Most habitats in the OCA are subjected to annual flooding. Although terrestrial receptors may occupy habitats in this area, residence time and exposure are reduced as a result of periodic inundation of habitats.

- The chemical mobility of lead is low and lead mobility is generally restricted to physical transport mechanisms. Some of the lead is enclosed in a glass matrix and the sorption capacity of soil and sediments for lead in non-vitrified forms is high because of the presence of iron and manganese hydrous oxides and sulfides, clays, and organic matter, which collectively bind lead.

- The bioavailability of lead to receptor species in soil, surface water, and sediments of the OCA is expected to be low because of the low solubility of forms of the lead and the presence of cations in soil that would control dissolution of lead minerals in the gastrointestinal tract.

- The size of affected habitat at the EFA and OCA is small relative to the home ranges of most ecological receptors, in particular predators and large herbivores. Because these individuals will spend only a portion of their time in the affected areas, the frequency, magnitude, and duration of potential exposure are likely to be low. In addition, the isolation of the sludge deposits below ground at the EFA minimizes or eliminates the potential for exposure to these media in this area.

- Food-web models developed using species-specific screening criteria and conservative exposure assumptions regarding lead bioavailability and bioaccumulation indicate that soil lead concentrations in the OCA do not represent a significant population-level risk to ecological receptors. However, a limited number of individual organisms of receptors with small home ranges restricted to the deltaic area, such as short-tailed shrews or white-footed mice, could potentially be affected by lead concentrations as food-web exposure models indicate hazard quotients above unity.

- The absence of stressed vegetation noted in both the EFA and the OCA during field reconnaissance surveys provides a qualitative line-of-evidence to suggest that elevated lead concentrations in soil and sediments in these habitats are not phytotoxic, although Site-specific phytotoxicity data are not available. In addition, the bioavailability of lead in soils to plants, and hence to plant-eating mammals, is expected to be low because lead does not readily translocate to plants.

- Toxicity testing in 2007 indicates that lead concentrations up to 3,180 mg/kg in sediment have no adverse effects on growth or survival of two invertebrate species, *Hyalella azteca* and *Chironomus tentans.*

The reader is referred to Appendix G for a complete discussion of the approach, analyses, discussion, and rationale supporting the Phase I ecological assessment and its conclusions.

8 Conclusions

This section summarizes the results of the remedial investigation for the Site. The remedial investigation is based on a synthesis of information and data from a number of sources and Site activities. These sources and activities include available records on facility operations, data from numerous previous investigations, and data collected during the remedial investigation, including supplemental sampling completed in 2003, 2005, and 2007. The remedial investigation was completed in accordance with the RI/FS work plan (PTI 1995), as amended (GE 1997; BBL 2002, 2003, 2006), and approved by OEPA. Conclusions of the remedial investigation with regard to the nature and extent of contamination, transport and fate processes, HHRA, and ecological assessment are summarized below.

8.1 Nature and Extent of Contamination

Information on the nature and extent of contamination at the Site was compiled from both data collected during the remedial investigation and from results of numerous previous investigations for the following areas of interest: EFA, East Swale, Onsite Soils, Adjacent Fields, former Oil Skimmer Pond, South Ditch, and OCA. The PCoIs originally identified for the Site (PTI 1995) included antimony, arsenic, barium, chromium, lead, nickel, fluoride, and petroleum hydrocarbons (as TPH). The historical and remedial investigation data confirm that PCoI metal concentrations in soil/sediment correlate well with lead concentrations, with higher PCoI levels associated with elevated lead concentrations. Based on remedial investigation results, certain PAHs have also been identified as PCoIs for the South Ditch where localized accumulations of oily sediment occur.

As described below and documented in this report, these areas of interest and PCoIs were screened to specific areas and CoIs for further detailed evaluations as part of the HHRA and the ecological assessment. For the HHRA, the media and areas of interest include sludge at the EFA and soil/sediment at the East Swale, South Ditch, and OCA. The CoIs for the HHRA include antimony, arsenic, and lead, with selected PAHs (as noted above) for the South Ditch. For the Phase I ecological assessment, the areas of interest include the EFA and OCA, and the CoI is lead.

8.1.1 East Fenced Area

The extent of glass fines (sludge) at the EFA was delineated by visual observation in backhoe scrapings and hand-dug pits and through collection and analysis of soil samples. The results indicate that the area of sludge deposits is slightly larger than previously estimated, with a limited region of mixed soil and sludge materials delineated to the east beyond the current EFA fenceline. The permeability of the sludge deposits is also higher than previously assumed, with the ability to transmit infiltrating water similar to a fine-grained silty sand. The available elevation data indicate that the sludge deposits are located above the shallow groundwater that underlies the EFA. Soil sampling on or near the former roadway on the west side of the EFA (in the vicinity of former Dames and Moore station 25D) confirmed that PCoI concentrations

are generally low and insignificant from a risk-based perspective. As discussed with OEPA, the historic TCLP sample data will not be the sole basis for any future waste characterization associated with any future proposed remedial actions at the Site.

Five quarters of groundwater sampling were conducted at the EFA during the remedial investigation (as discussed in the report, the fourth quarter data are not considered representative and were not used). Barium was the only inorganic PCoI detected during each quarter. Antimony and nickel were not detected, and lead was detected only once in an unfiltered sample (believed to be particulate detection). All detections were at low concentrations and were well below federal MCLs. The remedial investigation data suggest minimal, if any, effects of the sludge deposits on the shallow groundwater at the EFA. Considering the permeable nature of the sludge deposits, these data further confirm the immobility of the PCoIs in the subsurface environment at the EFA.

8.1.2 Adjacent Fields

Remedial investigation soil sampling in the area north of the plant confirmed historical information indicating that, while concentrations of lead and fluoride may be slightly elevated in surface soils, the concentrations are insignificant from a risk-based perspective. Additionally, as discussed with OEPA, the soils data from the Adjacent Fields was determined to appropriately represent background arsenic concentrations for this Site.

8.1.3 East Swale

PCoI concentrations, in particular lead, in East Swale soils/sediments are highest in the southern end of the swale near the former raw material handling areas. The concentrations decrease to the north along the swale; however, the range in lead concentrations remains similar along the area bordering the east end of the plant near the former cullet management areas. The distribution of elevated lead concentrations along the swale is generally restricted to the soil/sediment along the bottom of the swale. Finally, no samples were collected from the East Swale for TCLP analysis. However, as further discussed below, only one of the eleven samples collected from the South Ditch and OCA (where total lead concentrations are generally consistent with those observed in East Swale soils/sediments) had a TCLP lead concentration greater than the maximum allowable RCRA regulatory concentration of 5 mg/L.

The vertical profile of the PCoI metals in the swale is somewhat different from that observed for other areas, suggesting a complex history of filling or reworking of the materials in the swale. The highest groundwater levels observed in the vicinity of the swale are at least 6–11 ft below the affected soils/sediments along the bottom of the swale. The hydraulic and geochemical conditions indicate that PCoI-bearing materials in the swale are isolated and essentially immobile.

8.1.4 Former Oil Skimmer Pond

Field observations and borehole confirmation soil sampling beneath the former pond location indicate that adequate closure measures have been completed for the former pond. Additional backhoe and Geoprobe® sampling of a sandy horizon east of the former pond indicated that any residual hydraulic oil in the subsoils is limited in area and low in concentration; the limited residuals also do not contain hazardous constituents at levels of concern.

8.1.5 South Ditch

Results of both historical and remedial investigation soil/sediment sampling in the South Ditch indicate that elevated lead (and sometimes other inorganic PCoI) concentrations occur below the former East Swale outlet near outfall 19+30 and extend to the western boundary of the Site. The highest concentrations were found in the vicinity of the storm sewer outfalls and in localized sediment accumulation areas downstream of the outfalls. Similar to other Site areas, the elevated lead concentrations are present primarily in the surface soils/sediments. TPH and PAHs were detected in the South Ditch near the outfall at station 19+30, extending to the western boundary of the Site at a sediment accumulation area upstream of the culvert beneath the highway. However, the sediment data from the supplemental TPH investigation performed in 2005 indicated consistently lower levels of TPH in South Ditch sediments, with concentrations ranging from undetected to 250 ppm. PCBs were not detected in any sediment samples. Because PAHs are not expected in hydraulic oil and were not detected in the oily materials associated with the Former Oil Skimmer Pond, the PAHs found in the South Ditch soils/sediments (and possibly the bulk of TPH associated in these media) are believed to be related to discharges from the storm sewer outfall at location 19+30 on the ditch. Finally, two sediment samples were analyzed for TCLP lead during the remedial investigation. One sample did not contain lead at a detectable concentration and the other sample contained lead at a concentration less than the maximum allowable RCRA regulatory extract concentration of 5 mg/L. As discussed with OEPA, the historic TCLP sample data will not be the sole basis for any future waste characterization associated with any future proposed remedial actions at the Site.

Groundwater and surface water elevation data collected along the ditch, and beneath other areas of the Site, indicate that the shallow groundwater discharges to the South Ditch extending westward from the vicinity of the EFA. The shallow groundwater from beneath plant areas and surrounding agricultural fields provides a component of baseflow (recharge) to the ditch along this reach. Geochemical analyses and surface water sampling indicate that any waters in contact with the PCoI-bearing soils/sediments along the ditch do not cause dissolution and migration of PCoIs to any significant degree.

8.1.6 Offsite Creek Area

Elevated lead concentrations in the OCA are primarily found in depositional areas, where lead-bearing particulates were transported and locally accumulated, probably during major storm water runoff events. The highest lead levels (typically 1,000–10,000 mg/kg range) were found in a roughly triangular deltaic area where the offsite creek divides immediately west of the

railroad tracks. Elevated lead concentrations also occur along portions of the upper creek between Highway 23 and the railroad tracks. Together, these areas comprise approximately 2 acres (or less than 20 percent) of the OCA. Downstream from these areas, elevated lead concentrations are generally restricted to limited overbank soils/sediments, and an older abandoned channel to the south and east of the current farm ditch, which is typically dry. The elevated lead concentrations were highest in surface sediments and decrease rapidly with depth, with most occurring at depths of less than 1 ft, except in central sediment accumulation zones within the deltaic area and western portion of the upper creek.

With respect to TPH analyses, the sediment data from the supplemental TPH investigation performed in 2005 indicated low levels of TPH (ranging from 25 to 52 ppm) in OCA sediments located west of the railroad tracks. Finally, nine samples were collected from the OCA for TCLP analysis. Only one of the nine samples collected from the OCA had a TCLP lead concentration greater than the maximum allowable RCRA regulatory extract concentration of 5 mg/L. As discussed with OEPA, the historic TCLP sample data will not be the sole basis for any future waste characterization associated with any future proposed remedial actions at the Site.

Surface water sampling in the OCA confirms that lead is not dissolving and migrating from the OCA sediments or upstream sediment to any significant degree. Dissolved lead was not detected in the surface water samples and total lead concentrations (maximum concentrations less than 15 μg/L) were well below Ohio water quality standards and the allowable levels specified in Thomson's NPDES discharge permit.

8.1.7 Lead Particle Analyses

The lead mineralogy analyses indicate that 5–17 percent of lead in soils/sediments from the East Swale, South Ditch, and OCA is entrained in glass particles. This lead is generally present within small (less than 3 μm in diameter), high lead (i.e., 15.4 ± 3.2 weight percent) glass particles. Glass particles are present as angular to sub-angular shards, indicating that vitrified material in sediments has not weathered appreciably in the environment. The lead not entrained in glass is present as discrete particles of common soil alteration phases (e.g., sulfides, sulfates, oxides, and phosphates) and lead adsorbed on the surfaces of fine sediment particles (i.e., iron hydroxides, organic matter, silts, and clays). Although lead concentrations in the OCA sediments are approximately equal in the fine and coarse sediment fractions, lead mass is present primarily (88 percent on average) in the coarse sediment material (i.e., greater-than-500-μm size fraction).

8.2 Transport and Fate of Potential Chemicals of Interest

To evaluate PCoI transport and fate, the distribution of PCoIs in sludge, soils, sediments, surface water, and groundwater was interpreted with respect to past and present sources and potential transport processes.

8.2.1 Surface Water Transport

Based on the physical and chemical properties of the PCoIs, the surface water hydraulics and geochemical conditions at the Site, and the sampling results, particulate transport is the primary mechanism for PCoI migration in the environment. Due to the low solubility of the PCoIs and PCoI-bearing materials, dissolved-phase transport at the Site is minimal. Interactions between potential dissolved-phase PCoIs and soils/sediments in the environment at the Site result in the rapid precipitation of PCoIs into secondary mineral forms. Movement of suspended (or resuspended) particulates in surface water has allowed transport of PCoIs (principally lead) to and along portions of the South Ditch, and downstream of the Site to the OCA. The maximum lead levels (typically 1,000–10,000 mg/kg range) were found in the roughly triangular deltaic area where the offsite creek divides immediately west of the railroad tracks. Sampling results indicate that significant transport away from this area in the OCA is not occurring.

8.2.2 Groundwater Transport

The transport of PCoIs from Site soils, sediments, or sludge deposits to shallow groundwater is insignificant at the Site. Concentrations of PCoIs in groundwater, when detected, are below MCLs, even when monitoring is conducted adjacent to source materials (i.e., sludge deposits at the EFA) containing high concentrations of PCoIs. The lack of significant transport is due to both the low solubility of the PCoIs in the source material and the rapid binding of potentially dissolved PCoIs by adsorption or precipitation in soil/sediments at the Site.

8.3 Human Health Risk Assessment

The HHRA evaluated the potential for adverse human health effects to result from exposures to Site-related chemicals under current and potential future Site conditions in the absence of any action to control or mitigate exposures to these chemicals (i.e., under the no-action alternative). Scenarios for potential human exposure included the onsite occupational and recreational (or trespasser) scenarios. Additionally, at the request of OEPA, a hypothetical future residential scenario in the OCA was evaluated, with results presented in Appendix I. The risk evaluations included screening analyses based on conservatively derived risk-based concentrations, maximum observed Site concentrations, and patterns of Site use, with additional consideration of hypothetical future scenarios.

In assessing the potential effects of lead, this assessment found that all predicted 95th percentile blood lead concentrations calculated assuming a weekly exposure frequency were less than EPA's benchmark of 10 μg/dL for all receptors in all areas, with the single exception of a worker's exposure to EFA sludge using a contact-intensive soil ingestion rate (Tables 6-16, 6-18, 6-19, and I-4). For the onsite worker, all lead concentrations at any depth in the East Swale and South Ditch were associated with predicted 95th percentile blood lead concentrations below the EPA target when using the default soil ingestion rate (Table 6-16). When assuming a higher soil ingestion rate, concentrations in the East Swale and the South Ditch resulted in exceedances of the target blood lead level for the worker only when assuming daily exposures. Contacting EFA sludge at higher exposure frequencies (i.e., more than weekly) or when assuming a higher soil ingestion rate also yielded predicted blood lead levels that exceeded the

target blood lead level (Table 6-16); however, the EFA sludge is covered with 2 ft of soil, which has chemical concentrations below screening levels. Under the hypothetical assumption of future residential development within the OCA (presented in Appendix I), the modeled blood lead levels for young children were2.06–2.13 μg/dL, well below the regulatory benchmark of 10 μg/dL (Table I-3). Under the assumption of a residential older child visiting the deltaic or upper creek areas from a home within the OCA, modeled blood 95th percentile lead levels did not exceed the regulatory benchmark of 10 μg/dL assuming weekly visits (for any soil ingestion rate) or when using a lower soil ingestion rate (for any exposure frequency) (Table I-4). Some combinations of higher soil ingestion rates and/or more frequent exposures to the OCA yielded predicted 95th percentile blood leads above the target level (Table I-4). Similar results were obtained for the recreational user/trespasser scenarios in the South Ditch and OCA, which yielded predicted 95th percentile blood lead concentrations well below EPA's benchmark value assuming weekly visits (for any soil ingestion rate) or when using a lower soil ingestion rate (for any exposure frequency) (Tables 6-18 and 6-19). Some combinations of higher soil ingestion rates and/or more frequent exposures to the South Ditch or OCA yielded predicted 95th percentile blood leads above the target level for the recreational user/trespasser scenarios (Tables 6-18 and 6-19).

At the request of OEPA, the predicted blood lead concentrations for the onsite worker and recreational user/trespasser scenarios were compared to lower benchmark blood lead levels of 5, 6, 7, 8, and 9 μg/dL (Tables 6-22 through 6-24) to address the possibility that EPA may, in the future, target a 95th percentile blood lead level below the current level of 10 μg/dL. For the onsite worker, weekly exposure to lead concentrations in the East Swale and South Ditch result in only a single exceedance of the lowest benchmark of 5 μg/dL when using the contact-intensive soil ingestion rate (Table 6-22). For exposure frequencies greater than weekly for an onsite worker, particularly when combined with a higher soil ingestion rate, exceedances of some of the benchmark blood lead levels were seen for concentrations in the East Swale and South Ditch (Table 6-22). For exposures of an onsite worker to the EFA sludge, weekly exposures using the default soil ingestion rate did not exceed the benchmarks of 7, 8, or 9 μg/dL; however, all other scenarios result in exceedances of all five of these lower benchmarks. For the recreational user/trespasser, the results of the additional comparisons requested by OEPA indicate that, under the assumption of weekly exposures, there are no exceedances of benchmark blood lead levels of 7, 8, or 9 μg/dL in either the South Ditch (Table 6-23) or the OCA (Table 6-24), and exceedances of lower blood lead targets (5 or 6 μg/dL) occur only when incorporating the highest soil ingestion rate. Further, there are no exceedances of any benchmark under any scenario for the non-deltaic area of the OCA. However, the use of exposure frequencies greater than weekly for the South Ditch and other areas of the OCA resulted in exceedances of some of these lower blood lead benchmarks (Tables 6-23 and 6-24).

The risk estimates calculated for each area for carcinogenic CoIs at the Site are well within EPA's acceptable risk range and are less than OEPA's target level of 1×10^{-5} (Table 6-17). No carcinogenic risk estimates calculated for the Site exceed 1×10^{-4}. Arsenic was the only carcinogenic CoI identified for most of the areas of interest evaluated in the quantitative risk calculations. In the one case where other carcinogenic CoIs were identified for a Site area, arsenic remained the primary contributor to the estimated cancer risks. The total cancer risk estimated for the onsite worker scenario assuming contact with all three Site areas of interest is

6×10^{-6} if monthly contact is assumed and 2×10^{-5} if weekly contact is assumed (Table 6-17). Risks estimated for the recreational/trespasser scenarios in the South Ditch and OCA were approximately 2- to 3-fold less (Table 6-20).

In assessing the potential noncancer health effects associated with non-lead CoIs at the Site, the hazard indices calculated for the recreational/trespasser exposure scenarios are 1–2 orders of magnitude less than EPA's and OEPA's target hazard index of 1 (Table 6-20). The hazard indices calculated for the onsite occupational scenario are also 1–2 orders of magnitude less than EPA's and OEPA's target level for each of the three Site areas individually, and a factor of 5–20 less than the target level if it is assumed that an individual worker contacts all three areas (Table 6-17). The primary contributor to noncancer hazard indices is arsenic.

8.4 Phase I Ecological Assessment

A Phase I ecological assessment was completed to evaluate whether a Phase II ecological assessment is warranted. Habitats on and around the Site were evaluated with regard to habitat quality and extent, potential ecological receptors, PCoIs, and exposure pathways, and sediments from the OCA were tested in sediment toxicity bioassays. Key conclusions of the evaluation are that:

- Only terrestrial environments are of concern
- The EFA and OCA are the only areas of interest with habitats for consideration
- Lead is the only CoI for potential ecological receptors
- The only relevant exposure routes for the potential receptors are the food ingestion and incidental soil ingestion pathways.

The conclusion that environmental concentrations of lead at the EFA and OCA do not pose a significant ecological risk to receptor populations is based on the integration of several factors. These factors include the overall habitat quality and distribution of lead at the EFA and OCA, size of receptor home ranges relative to the size of the EFA and OCA, proportion of the receptor population potentially affected by exposure to lead, bioavailability of lead, toxicity of lead to potential receptors, and quantitative food-web exposure models for the most highly exposed receptors.

The conclusion of the Phase I ecological assessment is that a Phase II ecological assessment (i.e., a baseline ecological risk assessment) is not necessary. The areas of interest comprise only a minor portion of the home ranges of the larger receptors and thus the major exposure pathway (i.e., food ingestion) is unlikely to result in any significant exposure. For the EFA and OCA, the isolated media or small areas affected by elevated lead concentrations, and the effect of physical stressors on the OCA environment, argue against the potential for ecological risk. In addition, a review of the current data on geochemical characteristics controlling solubility and bioavailability of lead at the OCA support the conclusion that lead in the environment of the

EFA and OCA is not highly bioavailable or readily taken up and incorporated by plants and animals. Food-web models analyzed for small mammals at the OCA indicate that the likelihood of population level effects for these receptors is low. Toxicity tests indicate that the likelihood of adverse effects from lead in sediments of the OCA to benthic invertebrates is negligible. Thus, it is unlikely that lead concentrations in environmental media of these onsite or offsite areas of interest pose a significant ecological risk to receptors.

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