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Baseline Human Health Risk Assessment
Work Plan
Waste Water Treatment Plants 1 and 3
and Ash Pits 1 and 3

Plum Brook Ordnance Works
Sandusky, Ohio

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Table of Contents

Page

List of Tables iv
List of Figures iv
List of Acronyms v
1.0 Introduction 1-1
1.1 Facility Description 1-1
1.2 Background 1-2
1.2.1 Waste Water Treatment Plants 1-2
1.2.2 Ash Pit Areas 1-3
1.3 Groundwater Use and Site Use 1-4
1.4 Protocol for the Baseline Human Health Risk Assessment 1-5
1.5 Work Plan Organization 1-6
2.0 Data Evaluation 2-1
2.1 Data Sources 2-1
2.2 Sorting the Analytical Data 2-1
2.3 Evaluation of Data Quality 2-2
2.4 Identification of Chemicals of Potential Concern 2-3
2.4.1 Risk-Based Screening 2-3
2.4.2 Frequency of Detection 2-5
2.4.3 Comparison to Background 2-5
2.4.3.1 Background Screening of Inorganics 2-6
2.4.3.2 Statistical Data Set Testing of Inorganics 2-7
2.4.3.3 Treatment of Organics 2-7
2.5 Data Evaluation Summary 2-8
3.0 Exposure Assessment 3-1
3.1 Conceptual Site Exposure Model 3-1
3.1.1 Physical Setting 3-2
3.1.1.1 WWTP1 3-6
3.1.1.2 WWTP3 3-6
3.1.1.3 AP1 3-6
3.1.1.4 AP3 3-7
3.1.2 Contaminant Sources, Release Mechanisms, and Migration Pathways .. 3-8

Table of Contents *(Continued)*

Page

3.1.3 Receptors and Exposure Pathways 3-8
3.1.3.1 Groundskeeper 3-9
3.1.3.2 Indoor Worker 3-11
3.1.3.3 Construction Worker 3-12
3.1.3.4 On-Site Resident 3-13
3.1.3.5 Hunter 3-17
3.1.3.6 Hunter's Child 3-18
3.1.3.7 Other Receptors Not Considered 3-18
3.2 Quantification of Exposure-Point Concentrations 3-19
3.2.1 Soil, Groundwater, Surface Water, and Sediment Concentrations 3-19
3.2.2 Exposure-Point Concentrations in Air 3-21
3.2.2.1 COPC Concentrations from Dust 3-21
3.2.2.2 COPC Concentrations in Indoor Air 3-22
3.2.2.3 VOC Concentrations from Subsurface Soil in Ambient Air 3-29
3.2.2.4 Concentrations in Household Air from Groundwater Use 3-30
3.2.2.5 Concentrations of VOCs in Groundwater: Resident Dermal Uptake 3-31
3.2.2.6 Exposure-Point Concentrations of COPCs in Venison 3-31
3.3 Quantification of Chemical Intake 3-34
3.3.1 Inhalation of COPCs in Air 3-34
3.3.2 Incidental Ingestion of COPCs in Soil 3-34
3.3.3 Incidental Ingestion of COPCs in Sediment 3-35
3.3.4 Dermal Contact with COPCs in Soil, Sediment, or Water 3-35
3.3.5 Consumption of Venison 3-39
4.0 Toxicity Evaluation 4-1
4.1 Evaluation of Carcinogenicity 4-1
4.2 Evaluation of Noncarcinogenic Effects 4-4
4.3 Dermal Toxicity Values 4-5
4.4 Target Organ Toxicity 4-5
4.5 Sources of Toxicity Information Used in the Risk Assessment 4-6
5.0 Risk Characterization 5-1
5.1 Cancer Risk 5-1

Table of Contents *(Continued)*

Page

5.2 Noncancer Effects of Chemicals 5-3
6.0 Uncertainty Analysis 6-1
6.1 Types of Uncertainty 6-1
6.2 Sources of Uncertainty 6-2
7.0 Development of Risk-Based Remediation Criteria 7-1
8.0 Summary and Conclusions 8-1
9.0 References 9-1

List of Tables

Table	Title	Follows Tab
3-1	Receptor/Exposure Scenarios	
3-2	Variables Used to Estimate Potential Chemical Intakes and Contact Rates for Receptors	

List of Figures

Figure	Title	Follows Tab
1-1	PBOW Vicinity Map	
1-2	Locations of Areas of Concern	
3-1	Human Health Conceptual Site Exposure Model	

List of Acronyms

á	attenuation coefficient of VOC concentration between soil gas and indoor air
ABS	dermal absorption factor
AF	soil-to-skin adherence factor
AP	ash pit
AP1	Ash Pit 1
AP2	Ash Pit 2
AP3	Ash Pit 3
atm/m^3	atmospheres per cubic meter
BHHRA	baseline human health risk assessment
BSC	background screening concentration
BTEX	benzene, toluene, ethylbenzene, and xylenes
CDI	chronic daily intake
C_{source}	volatile organic compound concentration in soil gas at source of contamination
$C_{building}$	volatile organic compound concentration in indoor air in the building
cm	centimeter
cm^2	square centimeters
cm^2/second	square centimeters per second
cm^3/g	cubic centimeters per gram
COC	chemical of concern
COPC	chemical of potential concern
CSEM	conceptual site exposure model
DA	dose absorbed per unit body surface area per day
D&M	Dames and Moore, Inc.
DERP	Defense Environmental Restoration Program – Formerly Used Defense Sites
DM	dry matter
DNT	dinitrotoluene
DOE	U.S. Department of Energy
EPA	U.S. Environmental Protection Agency
EPC	exposure-point concentration
ET	exposure time
°F	degrees Fahrenheit
FI	fractional term
FS	feasibility study

List of Acronyms *(Continued)*

ft bgs	feet below ground surface
FUDS	Formerly Used Defense Sites
GAF	gastrointestinal absorption factor
gpm	gallons per minute
g/cm^3	grams per cubic centimeter
g/kg-day	grams per kilogram of body weight per day
g/m^2	grams per square meter
g/m^3	grams per cubic meter
HHEM	human health evaluation manual
HI	hazard index
HQ	hazard quotient
ILCR	incremental lifetime cancer risk
IRIS	Integrated Risk Information System
IT	IT Corporation
kg	kilogram
kg/day	kilograms per day
kg/m^2	kilograms per cubic meter
K_p	permeability coefficient
L/day	liters per day
LSI	limited site investigation
m^3	cubic meters
m^3/day	cubic meters per day
m^3/hr	cubic meters per hour
m^3/kg	cubic meters per kilogram
m/second	meters per second
MDC	maximum detected concentration
µg/L	micrograms per liter
$\mu g/m^3$	micrograms per cubic meter
mg/cm^2	milligrams per square centimeter
mg/m^3	milligrams per cubic meter
mg/day	milligrams per day
mg/kg	milligrams per kilogram
mg/kg-day	milligrams per kilogram per day
mg/L	milligrams per liter

List of Acronyms *(Continued)*

NASA	National Aeronautics and Space Administration
NCP	National Oil and Hazardous Substances Pollution Contingency Plan
OEPA	Ohio Environmental Protection Agency
PAH	polycyclic aromatic hydrocarbon
PBOW	Plum Brook Ordnance Works
PEF	particulate emission factor
QC	quality control
RBRC	risk-based remediation criteria
RBSC	risk-based screening concentration
RfC	reference concentration
RfD	reference dose
RI	remedial investigation
RME	reasonable maximum exposure
RSL	Oak Ridge National Laboratory – U.S. Environmental Protection Agency Regional Screening Level
SDWR	secondary drinking water regulation
SF	cancer slope factor
τ	time for concentration of contaminant in stratum corneum to reach steady state per event
Shaw	Shaw Environmental, Inc.
TNT	trinitrotoluene
TNTA	TNT Area A
TNTB	TNT Area B
TNTC	TNT Area C
UCL	95th percent upper confidence limit on the arithmetic mean
UF	uncertainty factor
USACE	U.S. Army Corps of Engineers
VOC	volatile organic compound
WRS	Wilcoxon Rank Sum
WWTP	waste water treatment plant
WWTP1	Waste Water Treatment Plant 1
WWTP2	Waste Water Treatment Plant 2
WWTP3	Waste Water Treatment Plant 3

1.0 Introduction

This baseline human health risk assessment (BHHRA) work plan was prepared to describe the protocol for evaluating human health risks associated with exposure to soil, groundwater, surface water, and sediment associated with Waste Water Treatment Plant 1 (WWTP1) and Waste Water Treatment Plant 3 (WWTP3) and Ash Pit 1 (AP1) and Ash Pit 3 (AP3) located at the former Plum Brook Ordnance Works (PBOW), Sandusky, Erie County, Ohio. This work is being conducted for the U.S. Army Corps of Engineers (USACE) under the Defense Environmental Restoration Program (DERP) - Formerly Used Defense Sites (FUDS). Investigations at PBOW under DERP-FUDS are being managed by the USACE Huntington District and technically overseen by the USACE Nashville District.

This work plan is consistent with U.S. Environmental Protection Agency (EPA) and Ohio Environmental Protection Agency (OEPA) guidance and with the procedures established in the BHHRA for TNT Areas A and C soil (IT Corporation [IT], 2001a), the BHHRA work plan for groundwater at PBOW (Shaw Environmental, Inc. [Shaw], 2005a) and the BHHRA work plan for Reservoir Number 2 Burning Ground (Jacobs Engineering Group, Inc., 2008).

1.1 Facility Description

PBOW operated from 1941 to 1945 as a manufacturing plant for 2,4,6-trinitrotoluene (TNT), dinitrotoluenes (DNT), and pentolite. Some of the areas used by the U.S. Department of Defense were decontaminated in the 1950s and 1960s; other areas have been decommissioned but not decontaminated. The PBOW facility is currently owned by the National Aeronautics and Space Administration (NASA) and is operated as the Plum Brook Station of the Lewis Research Center, which is headquartered in Cleveland, Ohio. In 1978, NASA declared approximately 2,152 acres of land as excess (IT, 1997). The Perkins Township Board of Education acquired 46 acres of the excess property for use as a bus transportation center. The Ohio National Guard has an agreement with the U.S. Army's General Services Administration to use 604 acres of the facility.

PBOW is located approximately 4 miles south of Sandusky, Ohio, and 59 miles west of Cleveland (Figure 1-1). Although located primarily in Perkins and Oxford Townships, the eastern edge of the facility extends into Huron and Milan Townships. PBOW is bounded on the north by Bogart Road, on the south by Mason Road, on the west by Patten Tract Road, and on the east by U.S. Highway 250. The areas surrounding PBOW are mostly agricultural and

residential. The facility is currently surrounded by a chain-link fence, and the perimeter is regularly patrolled. Access by authorized personnel is limited to established checkpoints. Public access is restricted, except during the annual deer hunting season.

1.2 Background

The 9,009-acre PBOW site was built in early 1941 as a manufacturing plant for TNT, DNT, and pentolite. Production of explosives began in December 1941 and continued until 1945. It is estimated that more than one billion pounds of explosives were manufactured during the 4-year operating period.

NASA acquired PBOW in 1963 and presently utilizes approximately 6,400 acres for conducting space research. The site is operated by NASA as the Plum Brook Station of the John Glenn Research Center, which is headquartered in Cleveland, Ohio. In 1978, NASA declared approximately 2,152 acres of land as excess (IT, 1997). The Perkins Township Board of Education acquired 46 acres of the excess property for use as a bus transportation center. The Ohio National Guard has an agreement with the U.S. General Services Administration to use 604 acres of the facility.

1.2.1 Waste Water Treatment Plants

Three waste water treatment plants (WWTP) at PBOW received waste water from TNT manufacturing operations. In addition to WWTP1 and WWTP3, Waste Water Treatment Plant 2 (WWTP2) received TNT manufacturing wastes. WWTP1 is located approximately 2,800 feet west of TNT Area A (TNTA) and received waste water from the settling basins located at TNTA and from TNT Area B (TNTB). WWTP2 is located approximately 2,000 feet northeast of the western portion of TNT Area C (TNTC) and received waste water from TNTC. WWTP3 is located between WWTP1 and WWTP2, approximately 2,800 feet north-northeast of TNTC and approximately 6,000 feet north-northwest of TNTB (Figure 1-2). The purpose of the plants was to reduce the volume of waste water discharged from each of the manufacturing areas to the West Area Red Water Ponds and Pentolite Road Red Water Pond. The waste water that was received from the TNT manufacturing areas consisted of spent sulfuric and nitric acids and red water from the TNT purification process. Chemicals in the waste streams included sodium salts of sulfite, sulfate, nitrite, nitrate, sulfonates of unwanted TNT isomers, trinitrobenzoic acid, trinitrobenzaldehyde, trinitrobenzyl alcohol, nitrotoluenes, and dinitrotoluenes (Dames and Moore, Inc. [D&M], 1997).

Each of the WWTPs consisted of a neutral storage tank, an equalization tank, a condensate storage tank, an evaporator building, an incinerator, and a thick liquid storage tank. WWTP1 also included a caustic storage tank and a raw waste storage tank. The plants received waste water from the TNT manufacturing settling basins and neutralized the slurry through a chemical depuration process. The liquid was discharged to open ditches or ponds surrounding the facilities or was thickened by evaporation. The thickened liquid was then incinerated. The incinerators were located to the east of the storage tanks at WWTP1 and to the north of the storage tanks at WWTP3. Ash from the incinerator was disposed of in nearby ash pits (AP) associated with the WWTPs (USACE, 2000a).

1.2.2 Ash Pit Areas

As noted previously, PBOW was built in early 1941. TNT, DNT, and pentolite were manufactured at PBOW until 1945. Three power stations, Power House 1, Power House 2, and Power House 3, were constructed and utilized to support the TNT manufacturing process. Each power station consisted of a main power house, a coal storage area, and an aboveground fuel storage tank. The fuel storage tank was surrounded by a berm to contain any potential spills or leaks. Each power house building consisted of a boiler house, compressor room, electrical room, filter room, and locker room. The buildings also contained two to four large coal-burning boilers, a turboelectric generator, a feed water treatment system, and several steam-driven or electric air compressors. The generated steam was used for space heating, driving compressors, and generating electrical power. Coal ash generated from each of the boilers in the power house was collected in pits. Water was added to the ash, producing a slurry that flowed through a sluice trench to an ash sump located at the end of each power house. From the ash sump, the ash slurry traveled through a pipeline to a nearby AP (USACE, 2000b). Figure 1-2 shows the locations of the three AP areas on PBOW property. Based on topographical quadrangles (dated 1959 and 1969), aerial photographs, and a visual site survey conducted in 1999, the AP areas are noted to have essentially remained unchanged. An exception to this is that AP1 has become overgrown with thick vegetation.

AP3 is located approximately 700 feet southwest of the intersection of Maintenance and Ransom Roads and is west of the former NASA K-Site Test Facility research building (former Power House 3) (Figure 1-2). During its operation, the K-Site supplied noncontact cooling water to AP3. It reportedly held water during this time, although the depth of the water was unknown (USACE, 2000b). A pipe at the northeastern corner of the pit supplied the cooling water. It is possible that this is the original pipe that once supplied the ash slurry to the pit. Boundaries of the pit appear to be the same as the original boundaries, with the exception of an indention to the

west on the eastern boundary (USACE, 2000b). Operations at the K-Site were officially abandoned in 2007; however, it is possible that the facility may be used for test programs in the future. Abandoned railroad tracks running in a north-south direction are located immediately east of AP3. The pit is partially surrounded by thick vegetation, with mature and smaller trees. The original surface water impoundment was intact as of 1999 (USACE, 2000b), holding water supplied by NASA's K-Site Test Facility. Water was discharged from the former pond by means of an east-west-trending drainage ditch that eventually discharges into Pipe Creek (USACE, 2000b). Since the K-Site ceased operations, apparently the major source of water to AP3 has been eliminated. The presence/lack of standing water associated with various site visits and recent aerial photographs are described in further detail in Section 3.1.1.4.

1.3 Groundwater Use and Site Use

Two groundwater aquifer systems are utilized for drinking water in the area, a carbonate aquifer to the west and a shale aquifer to the east (Shaw, 2005b). PBOW is located within the transition of the two systems. Upwards of 170 private drinking water wells permitted by the Erie County Health Department are located within 4 miles of PBOW. Permits are not required for agricultural wells. The Erie County Health Department does not permit using surface water as private drinking water. Lake Erie and Sandusky Bay are used for recreational swimming, fishing, and boating. A shallow groundwater system within the unconsolidated material atop the bedrock exists under much of the site.

In this BHHRA work plan, the term "facility" refers to the entire former PBOW property, and the term "sites" refers to areas within PBOW under investigation, in this case, WWTP1, WWTP3, AP1, and AP3. Current site use of the PBOW facility is classified as industrial for the purpose of identifying plausible human receptors and exposure pathways for evaluation in the BHHRA. D&M (1997) describes potential future uses of all or portions of the facility as follows:

- Continued industrial use (NASA activities and programs).
- Recreational use of portions of the site by hunters and fishermen.
- Portions of the site may be sold to state or local government or private individuals (no land use restrictions were mentioned).
- Parts of the facility may be used in the future for residential or agricultural purposes.

- Parts of the facility may be used for training by the National Guard.
- Construction activities may be performed during development of any of the sites.

In summary, future site use of WWTPs 1 and 3 and APs 1 and 3 is considered to be industrial or residential for the purpose of developing receptor and exposure scenarios. Even though hunting is not currently permitted in any of these four WWTP and AP sites, hunting is permitted in other areas within PBOW; therefore, future use of these WWTP and AP sites for hunting is evaluated in this BHHRA It is assumed that groundwater may be developed as a source of potable water in the future. Refer to Section 3.1.3 for a discussion of receptors and exposure scenarios.

1.4 Protocol for the Baseline Human Health Risk Assessment

The purpose of this work plan is to describe the protocol for evaluating risk to human health at WWTP1, WWTP3, AP1, and AP3. This work plan is intended to serve as the template for the BHHRA report. A BHHRA is a stand-alone document, chapter or section; i.e., all the equations and values necessary for quality control (QC) and replication of computations must be contained within the report itself.

The work plan is based on EPA, USACE, and OEPA guidance, including, but not limited to, the following:

- U.S. Environmental Protection Agency (EPA), 1989a, ***Risk Assessment Guidance for Superfund***, Volume I, Human Health Evaluation Manual (Part A), Interim Final, Office of Emergency and Remedial Response, Washington, D.C., EPA/540/1-89/002.
- U.S. Environmental Protection Agency (EPA), 1991a, ***Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual Supplemental Guidance, Standard Default Exposure Factors***, Interim Final, Office of Solid Waste and Emergency Response, OSWER Directive: 9285.6-03.
- U.S. Environmental Protection Agency (EPA), 1992a, ***Guidance on Risk Characterization for Risk Managers and Risk Assessors***, Memorandum from F. Henry Habicht II, Deputy Administrator, to Assistant Administrators, Regional Administrators, February 26.
- U.S. Environmental Protection Agency (EPA), 1997a, ***Exposure Factors Handbook***, Office of Research and Development, National Center for Environmental Assessment, Washington, D.C., August, EPA/600/P-95/002Fa.
- U.S. Army Corps of Engineers (USACE), 1999, ***Risk Assessment Handbook, Volume I: Human Health Evaluation***, Engineer Manual EM 200-1-4.
- U.S. Environmental Protection Agency (EPA), 2002, ***Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites***, Office of Solid Waste and

Emergency Response, Washington, D.C., 9355.4-24, December.

- U.S. Environmental Protection Agency (EPA), 2004a, ***Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual (Part E - Supplemental Guidance for Dermal Risk Assessment)***, Final, Office of Superfund Remediation and Technology Innovation, Washington, D.C., EPA/540/R-99/005, July.

- U.S. Environmental Protection Agency (EPA), 2009a, ***ProUCL Version 4.00.04***, Office of Research and Development, Technology Support Center Characterization and Monitoring Branch, Las Vegas, Nevada, February, on line at http://www.epa.gov/esd/tsc/form.htm.

- U.S. Environmental Protection Agency (EPA), 2009b, ***ProUCL Version 4.00.04 Technical Guide***, Draft, Office of Research and Development, Technology Support Center Characterization and Monitoring Branch, Las Vegas, Nevada, February, EPA/600/R-07/041.

- U.S. Environmental Protection Agency (EPA), 2009c, ***ProUCL Version 4.00.04 User Guide***, Draft, Office of Research and Development, Technology Support Center Characterization and Monitoring Branch, Las Vegas, Nevada, April, EPA/600/R-07/038.

It should be noted that the protocol presented herein may differ slightly from that used in previous BHHRAs as a result of updated risk assessment guidance and ongoing communication with OEPA, the primary regulatory authority for PBOW. The differences represent refinements or upgrades, particularly regarding levels of documentation that were not available for the earlier BHHRAs. Their inclusion at this point in time does not imply that the earlier BHHRAs are deficient or that substantially different conclusions would be drawn if they were redone using the present protocol.

Ideally, this work plan captures and solidifies all details of the protocol for BHHRAs at WWTPs 1 and 3 and APs 1 and 3. However, BHHRA knowledge is dynamic, and improvements and refinements occur frequently. Therefore, both USACE and OEPA reserve the right to initiate discussion regarding future changes to the protocol. The need for change is a matter of professional judgment, depending in part on the effect of the proposed change on the projected outcome or conclusions of the BHHRA and the cost of changing the protocol.

1.5 Work Plan Organization

The remainder of this document is organized as follows:

- ***Chapter 2.0, Data Evaluation.*** Identifies data sources, evaluates data quality, identifies chemicals of potential concern (COPC), and provides a background screening and evaluation protocol.

- ***Chapter 3.0, Exposure Assessment.*** Presents a conceptual site exposure model (CSEM), including contaminant sources, contaminant release mechanisms, receptors, and exposure pathways; describes exposure-point concentrations (EPC); and presents methods for calculating chemical intake and contact rates.

- ***Chapter 4.0, Toxicity Evaluation.*** Describes the potential for cancer and/or noncancer human health effects, provides an estimate of the quantitative relationship between the magnitude of dose or contact rate and the probability and/or severity of adverse effects, identifies the toxicity values that are used in the BHHRA, and describes the development of dermal toxicity values.

- ***Chapter 5.0, Risk Characterization.*** Combines the output of the exposure assessment and toxicity assessment to quantify the risk to each receptor at each site. Risks associated with exposure to all appropriate media for each site will be evaluated.

- ***Chapter 6.0, Uncertainty Analysis.*** Identifies uncertainties in all phases of the BHHRA and discusses their individual effects on the risk assessment results, focusing on those issues that are most likely to have the greatest effect on risk estimates and/or risk management decisions.

- ***Chapter 7.0, Development of Risk-Based Remediation Criteria.*** Provides risk-based remediation criteria (RBRC) based on the methodology of the BHHRA. RBRCs are intended for consideration in the development of cleanup goals during the feasibility study (FS) process.

- ***Chapter 8.0, Summary and Conclusions.*** Provides a brief summary of the BHHRA, including quantitative results, uncertainties, and pertinent site information. Summary and discussion is focused on those results and issues that are most likely to directly affect site management decisions.

- ***Chapter 9.0, References.*** Presents the references used in the preparation of this document.

2.0 Data Evaluation

Data evaluation consists of a description of the appropriate data sources for each environmental medium sampled for each site, a discussion of data quality, a description of the methodology used for identification of the COPCs, and a summary of the COPCs for WWTP1, WWTP3, AP1, and AP3 environmental media.

2.1 Data Sources

All analytical data used in the BHHRA will be presented in a sample summary table and described as necessary in the BHHRA text. This data will include validated surface soil, subsurface soil, surface water, sediment, and monitoring well samples. The sample summary table will identify each sample used in BHHRA and the associated analytical suite. This will include samples collected as part of the limited site investigation (LSI) for WWTP1 and WWTP3 (USACE, 2000a), the LSI for AP1 and AP3 (USACE, 2000b), and the remedial investigation (RI) samples described in the Phase 1 (Shaw, 2008a) and Phase 2 (Shaw, 2009) RI work plans.

2.2 Sorting the Analytical Data

Prior to initiation of a BHHRA, a database of chemicals present in site samples will be compiled. This database includes all chemicals detected in any site medium. The data for each chemical will be sorted by medium. Surface soil and subsurface soil are considered separate media. Surface and subsurface soil data are combined to assess exposures under the construction worker and residential site-use scenarios, which would involve excavation and mixing of surface and subsurface soil. Combined surface and subsurface soil data are termed "total soil" in the BHHRA. The total soil COPC list is created by combining the list of COPCs identified in surface and subsurface soil. Thus, if a chemical is either a surface soil COPC or a subsurface soil COPC (or both), then that chemical is a total soil COPC.

Ideally, surface soil should be defined as samples taken from 0 to 1 foot below ground surface (ft bgs), and subsurface soil should be defined as samples taken from within the 1 to 10 ft bgs interval. The Phase 1 RI work plan for these sites (Shaw, 2008a) prescribes that surface soil samples be collected from 0 to 1 ft bgs, as possible. The LSI historical data appear to have been collected from a depth of either 0 to 1 or 0.5 to 1 ft bgs, which is consistent with the RI work plan. The RI work plan prescribes that subsurface samples be collected at depth of 3 to 5 and 8 to 10 ft bgs as possible. Note that if refusal or the water table is encountered before a depth of 10 ft bgs, samples from the deepest 2-foot interval are used.

2.3 Evaluation of Data Quality

The quality of the analytical data will be evaluated to select data for inclusion in the BHHRA. Data quality is expressed by the assignment of qualifier codes during the analytical laboratory QC process or during third-party data evaluation. Some of the more common qualifiers and their meanings are as follows (EPA 1989a):

U - Chemical was analyzed for but not detected; the associated value is the sample quantitation limit.

J - Value is estimated, usually below the reporting limit.

N - The analysis indicates an analyte for which there is presumptive evidence to make a tentative identification.

NJ - The analysis indicates a "tentatively identified analyte" and the reported value represents its approximate concentration.

UJ - The analyte was not detected above the reporting limit. However, the reporting limit is approximate and may or may not represent the actual limit of quantitation necessary to accurately and precisely measure the analyte in the sample.

R - QC indicates that the data are unusable (chemical may or may not be present).

B - Inorganic chemicals: the concentration is less than the reporting limit but greater than the instrument detection limit. Organic chemicals: the concentration in the sample is not sufficiently higher than concentration in the blank, using the 5-times, 10-times (5x, 10x) rule: A chemical is considered a nondetect unless its concentration exceeds five times the blank concentration. For common laboratory contaminants (acetone, 2-butanone [methyl ethyl ketone], methylene chloride, toluene, and the phthalate esters), the sample concentration must exceed 10 times the blank concentration to be considered a detection.

"J," "N," and "NJ" qualified data and "B" qualified inorganic chemical data are treated in the BHHRA as detected concentrations; "R" data and "B" qualified organic chemical data are not. "U" qualified data (nondetects) are treated in the BHHRA as nondetections. The use of data with other, less common qualifiers is evaluated on a case-by-case basis. Generally, data for which the identity of the chemical is unclear are not used in the BHHRA. If confidence is reasonably high that the chemical is present, but the actual concentration is somewhat in question, the data generally are used in the BHHRA.

Some chemicals may be analyzed under two different analytical programs. For example, the DNT isomers are analyzed by EPA Method 8330 for nitroaromatics as well as EPA Method 8270C for semivolatile organic compounds. Risks associated with the reported values from both analyses are considered in the risk characterization (Chapter 5.0) and discussed as appropriate in the uncertainty analysis (Chapter 6.0) of the BHHRA, together with potential issues such as the relative sensitivities (i.e., differences in respective reporting limits) of the methods.

2.4 Identification of Chemicals of Potential Concern

A screening process is used to identify COPCs, which are the detected analytes carried through the full risk assessment process. The objectives of COPC screening are to focus the risk assessment on those chemicals that may contribute significantly to overall risk and to remove from quantification those chemicals whose contribution is clearly inconsequential. COPC screening includes a risk-based screen which also considers status as a human nutrient (Section 2.4.1), a frequency-of-detection evaluation (Section 2.4.2), and a background screen (Section 2.4.3).

2.4.1 Risk-Based Screening

In the risk-based screen, the maximum detected concentration (MDC) is compared to the appropriate risk-based screening concentration (RBSC) for each analyte. The units of the MDC and RBSC are the same for each chemical in a given medium. In groundwater, for example, both the MDC and RBSC have units of micrograms per liter (µg/L) in water.

If the MDC of a chemical is less than or equal to its RBSC, then the chemical in this medium is not considered further in the BHHRA, because it is very unlikely that chemical concentrations at or below the RBSC would contribute substantially to risk. An analyte may be identified as a COPC if its MDC exceeds its RBSC. As indicated in Section 2.4, actual status as a COPC also depends on a chemical's frequency of detection (Section 2.4.2), concentration with respect to background (Section 2.4.3), and potential status as a nutrient. Groundwater RBSCs used in the BHHRA are derived from the Oak Ridge National Laboratory – EPA Region Screening Levels (RSL) table "tap water" values and RBSCs for soil are derived from "residential soil" RSL values (Oak Ridge National Laboratory-EPA, 2009). This is a change in the source of the RBSCs based on discussion between USACE and OEPA (2009). Previously, the groundwater and soil RBSC were derived from the corresponding EPA (2004b) Preliminary Remediation Goals. The soil RBSCs are applied to both surface and subsurface soil.

RSL values are based on a concentration equal to either an incremental lifetime cancer risk (ILCR) of 1E-6 or a noncancer hazard quotient (HQ) of 1, the threshold at (or below) which adverse noncancer effects are regarded as unlikely to occur. For the BHHRA, the noncancer values listed in the RSL tables are multiplied by a factor of 0.1 to provide additional protection for simultaneous exposure to multiple chemicals (EPA, 2004c). This results in RBSC values associated with an HQ of 0.1. For cancer risk, the RSL values are used directly as RBSCs in the BHHRA, because these values are based on an ILCR of 1E-6. The National Oil and Hazardous Substances Pollution Contingency Plan (NCP) identifies acceptable exposure levels that are generally associated with concentration levels that represent an excess upper bound lifetime cancer risk to an individual of 1E-6 to 1E-4 (EPA, 1990). This range is hereinafter referred to as the "NCP risk management range." Cancer risks associated with RSL values represent the lower end of this range. The RBSC for a chemical that elicits both cancer and noncancer health effects is selected based on either a cancer risk of 1E-6 or an HQ of 0.1, whichever associated concentration is lower.

The groundwater and soil RBSCs are derived from RSLs as described in the preceding paragraphs. The routes by which receptors may be exposed to sediment (i.e., incidental ingestion, dermal contact) are similar to those by which receptors may be exposed to soil. However, sediment contact is expected to be appreciably less intense than soil contact, due to the lower amount of time and frequency that one would be expected to be potentially in contact with sediment as compared with soil. Therefore, the soil RBSCs are increased by an order of magnitude prior to application in the BHHRA as sediment RBSCs. Similarly, surface water exposure is expected to be much less intense than exposure to groundwater; for instance, the exposure frequency is expected to be much lower for surface water and the incidental ingestion of surface water would be much lower than the ingestion of groundwater from the tap. For this reason, the groundwater RBSCs are multiplied by an order of magnitude prior to their use as surface water RBSCs. Even though the sediment and surface water RBSCs are an order of magnitude higher than the respective soil and groundwater RBSCs, these sediment and surface water RBSCs are regarded as protective of sediment and surface water receptors for screening because of the lower exposure rate to these media.

There are no RSLs for lead or sulfate in tap water, but the EPA (2006) Office of Water lists a treatment technique action level of 0.015 milligrams per liter (mg/L) for lead and a health-based advisory level of 500 mg/L for sulfate. These values will be used as RBSCs. It is noted that a secondary drinking water regulation (SDWR) of 250 mg/L exists for sulfate. SDWRs are non-

promulgated values, based on aesthetic characteristics, which are used as guidelines for public water systems.

The evaluation of essential nutrients is a special form of risk-based screening applied to certain ubiquitous elements that are generally considered to be required human nutrients. Essential nutrients such as calcium, chloride, iodine, magnesium, phosphorous, potassium, and sodium are generally considered innocuous at levels found in environmental media. There are no RSLs listed for these nutrients. Should any of these chemicals be identified as site-related, an exposure analysis will be performed whereby a daily dose of chemical from ingestion of the medium in question is calculated. The dose will be compared with levels known or expected to be safe or toxic, and/or with recommended daily allowances, depending on the availability of data.

2.4.2 Frequency of Detection

When confidence is high that a given chemical is present, the data generally are used in the BHHRA. For most chemicals, their detection is presumptive evidence of their presence. As suggested by EPA (1989a), chemicals that are reported infrequently may be artifacts in the data that do not reflect the actual presence of the chemical in question. For the BHHRA, chemicals that are reported only at low concentrations in less than 5 percent of the samples from a given medium will be excluded from further consideration, unless the presence of a given chemical is expected based on historical information about the site. Chemicals detected infrequently at high concentrations may identify the existence of contaminant plumes or limited "hot spots" and will be retained as COPCs.

2.4.3 Comparison to Background

A number of the chemicals detected in PBOW environmental media may have MDCs that exceed RBSCs but are part of normal background concentrations. Such chemicals may include inorganics and polycyclic aromatic hydrocarbons (PAH), a class of organic compounds which form from natural or anthropogenic combustion of organic matter, including fossil fuels, and are generally ubiquitous in the environment. Airborne PAHs associated with non-Department of Defense sources may be deposited on soil and leach to groundwater. Benzene, toluene, ethylbenzene, and xylenes (BTEX) compounds, as well as PAHs, may also be associated with background concentrations due to the presence of natural petroleum-derived compounds present in the vicinity of PBOW (see Section 3.1.1).

Site concentrations of inorganic chemicals in site environmental media may be compared to those of PBOW background using a two-step approach: 1) background screening and 2)

statistical data set testing. This second step (Section 2.4.3.2) is initiated only in cases where the concentration used for background screening is exceeded (refer to Section 2.4.3.1) and will be performed after the risk characterization (Chapter 5.0) and the results will be discussed in the uncertainty analysis (Chapter 6.0). No suitable background data set exists for overburden wells, so no background screening or statistical comparisons to background concentrations will be made for overburden groundwater samples. Similarly, no background screening is performed for surface water or sediment analytical data, as these media lack PBOW background data sets.

Inorganics and organics will be treated similarly from a quantitative perspective. However, all organics not eliminated on the basis of RBSC exceedance (Section 2.4.1) or infrequent detection (Section 2.4.2) will be carried through the risk calculation process (exposure assessment, toxicity assessment, and risk characterization). As presented in Section 2.4.3.3, organic compounds will be quantitatively eliminated as background related only through the uncertainty analysis (Chapter 6.0) of the BHHRA.

2.4.3.1 Background Screening of Inorganics

Background screening is applied to each inorganic whose MDC in soil or limestone bedrock groundwater exceeds the RBSC and that cannot be characterized as an infrequently detected analyte. In background screening, the MDC is compared to the PBOW chemical-specific background screening concentration (BSC). The background data set and derivation of soil BSCs for all PBOW soil investigations are described in IT (1998), and the background data set and derivation of BSCs for PBOW groundwater are described in the 2004 groundwater report (Shaw, 2005b). The background soil samples were collected from near the property boundary, away from any potential source areas, and the background groundwater wells were installed in off-site areas upgradient of PBOW sources. Briefly, BSCs were calculated for use at PBOW based on concentrations found in these background soil and bedrock monitoring well samples. Each BSC is either the MDC or the calculated 95th percent upper tolerance limit of the background data set, whichever value is lower (IT, 1998; Shaw, 2005b). The background monitoring well samples were collected using low-flow samples and were unfiltered.

The background screening consists of comparing the MDC of the site data set to the BSC. The chemical is regarded as a COPC if its MDC exceeds the BSC for that chemical or if no BSC could be determined due to a lack of detections in the background data set. COPCs are fully evaluated in the exposure assessment, toxicity assessment, and risk characterization. An inorganic analyte is not regarded as a COPC if its MDC is equal to or less than the BSC.

2.4.3.2 Statistical Data Set Testing of Inorganics

Statistical testing is performed to compare data sets of site inorganics data against the appropriate PBOW background data sets. As described in Section 2.4.3.1, the background data set for groundwater is found in the 2004 groundwater report (Shaw, 2005b), and the background data set for soil is found in the site investigation for the acid areas (IT, 1998). As mentioned above, background data sets do not exist for overburden groundwater, surface water, or sediment; therefore, a statistical background evaluation for COPCs in these media cannot be performed.

The method for statistical comparison of the site data sets to the background data sets, described in Appendix M of Shaw (2005b), is the Wilcoxon Rank Sum (WRS) statistical test (also known as the Mann-Whitney U test). WRS testing is performed for inorganics having MDCs that exceed the respective BSCs and are identified as COPCs based on RBSC comparison (Section 2.4.1) and frequency of detection (Section 2.4.2). All COPCs are carried through the risk characterization process; thus, statistical testing results are not used to screen out any chemicals.

Site data sets are interpreted as being significantly different from PBOW background if the associated p-level is less than 0.05. WRS statistical output and box-and-whisker plots of the various inorganic COPC data sets will be appended to the BHHRA for each inorganic data set evaluated against the appropriate site background data set; the WRS results will be discussed as part of the uncertainties associated with site relatedness. Analytes shown by the WRS results to exceed background (or for which the WRS testing was not run) are assumed to be site related, unless a qualitative chemical-specific explanation is presented in the uncertainties analysis as to why the analyte should not be regarded as site related. The WRS test will not be run in the BHHRA if the COPC was not detected in the PBOW background data set. Data sets for which the WRS results do not suggest site relatedness (i.e., site data and background data are not statistically different) are still evaluated for risks and hazards in the risk characterization (Chapter 5.0). The status of such compounds is discussed in the uncertainty analysis of the BHHRA (Chapter 6.0).

2.4.3.3 Treatment of Organics

As mentioned in Section 2.4.3, certain organic compounds (BTEX and PAHs) in site media may be attributable to background conditions. The MDCs of PAH and BTEX data may also be compared to BSCs (Section 2.4.3.1) and may be compared to PBOW background data using the WRS test (Section 2.4.3.2), but no organic compounds will be summarily screened out. Instead, all detected organic compounds are carried through the risk assessment process (i.e., exposure assessment, toxicity assessment, risk characterization) unless screened out on the basis of

comparison to RBSCs (Section 2.4.1) or characterized as infrequently detected (Section 2.4.2). Background contribution of organics will be discussed in the uncertainties analysis of the BHHRA, as applicable.

2.5 Data Evaluation Summary

Tables with the following information for each detected chemical, in each environmental medium, within each site will be prepared:

- Chemical name
- Frequency of detection
- Range of detected concentrations
- Range of detection limits
- Arithmetic mean of site concentrations
- 95th percent upper confidence limit on the arithmetic mean (UCL)
- Appropriate RBSC
- Appropriate BSC
- Selection/exclusion of chemical as a COPC.

Note that the estimation of the UCL values, provided for the COPCs, is discussed in Section 3.2.1. For reasons discussed in Section 3.2.1, nondetects with method detection limits greater than the MDC will not be included in the data set used to calculate the EPC (EPA, 1989a); however, the information from these high nondetects will be included in the rest of the data summary information (e.g., frequency of detection, range of detection limits). If any sample results are eliminated based on high method detection limits, the eliminated results will be identified in the data evaluation and discussed in the uncertainty analysis of the BHHRA. For duplicate samples, the associated values will be averaged in the data summary, if both samples are detects or if both are nondetects; if only one of the duplicates is a detect, then this detected value will be used in the data summary.

Analogous summary tables for the overburden groundwater samples collected during the RI using direct-push sampling methodology will be appended to the BHHRA. Note that direct-push groundwater samples are collected for nature-and-extent purposes to determine groundwater flow direction and the placement on monitoring wells. These data are not sufficiently representative of groundwater conditions for use in risk assessment. Therefore, they are not used to identify COPCs and are not quantitatively evaluated in the BHHRA. These direct-push sample groundwater summary tables will be appended only to provide ancillary information.

3.0 Exposure Assessment

Exposure is the contact of a receptor with a chemical or physical agent. An exposure assessment estimates the type and magnitude of potential exposure of a receptor to COPCs found at or migrating from a site (EPA, 1989a). An exposure assessment includes the following steps:

- Characterize the physical setting.
- Identify the contaminant sources, release mechanisms and migration pathways.
- Identify the potentially exposed receptors.
- Identify the potential exposure pathways.
- Estimate exposure concentrations.
- Estimate chemical intakes or contact rates.

The BHHRA described in this work plan for WWTP1, WWTP3, AP1, and AP3 will characterize potential exposures to COPCs in soil, groundwater, surface water, and sediment (if present) associated with their respective site areas. Separate evaluations of potential exposures at each of the four sites will be performed. Estimation of risk from potential exposure will be described separately for each site in the BHHRA risk characterization for each medium. Therefore, the respective CSEMs described in Section 3.1 include all environmental media evaluated for these four sites.

3.1 Conceptual Site Exposure Model

The CSEM provides the basis for identifying and evaluating the potential risks to human health in the BHHRA. The CSEM includes the receptors appropriate to all plausible site-use scenarios and the potential exposure pathways. The CSEM is graphically depicted on Figure 3-1. This presentation of all possible pathways by which a potential receptor may be exposed, including all sources, release and transport pathways, and exposure routes, facilitates consistent and comprehensive evaluation of risk to human health and helps to ensure that potential pathways are not overlooked. The elements of a CSEM include the following:

- Source
- Source media (i.e., initially contaminated environmental media)
- Contaminant release mechanisms
- Contaminant transport pathways
- Intermediate or transport media
- Exposure media
- Receptors
- Routes of exposure.

Contaminant release mechanisms and transport pathways are not relevant for direct receptor contact with a contaminated source medium (e.g., ingestion or dermal contact).

The receptors and pathways on Figure 3-1 reflect plausible scenarios developed from information regarding site background and history, topography, climate, and demographics as presented by D&M (1997) and the sitewide groundwater investigation (IT, 1997). Asterisks identify exposure pathways that are complete and addressed in the BHHRA. Justification for exclusion of other pathways is provided in the footnotes or the descriptions of the receptors in Section 3.1.3.7. No current or future exposures by off-site residents will be evaluated. The majority of the off-site residents are serviced by municipal water (from surface water sources). Although there are numerous private groundwater wells in the vicinity, including eight within 1 mile of the facility boundary, none of these are used as a potable source. Based on the investigations of other PBOW sites, natural hydrocarbons and hydrogen sulfide are known to be present within the bedrock limestone, and shale formation groundwater generally provides low yields and is of low quality (e.g., Shaw [2008b]); however, the groundwater underlying these sites cannot be summarily excluded for consideration as a tap water source based on natural water quality parameters. Therefore, given the presence of numerous off-site wells and the assumption of unrestricted future land use on site, the development of groundwater for on-site residential (or on-site worker) use as tap water is regarded as plausible for purposes of this work plan. If groundwater data and information collected during the RI indicate that potable use of either of these units is not plausible, that information will be presented in the BHHRA report.

3.1.1 Physical Setting

Climate/Meteorology. The climate in the Sandusky area is continental and strongly affected by Lake Erie. July is generally the warmest month (average high and low temperatures of 82 and 65 degrees Fahrenheit [°F], respectively), and January is generally the coldest (average high and low temperatures of 32 and 19°F, respectively) (The Weather Channel, 2004). On average, the first freezing day (low of 32°F or less) occurs in late October (average of 3 per month), and the last freezing day falls in early May (average of 1 per month) (National Oceanic and Atmospheric Administration, 1990). The average annual precipitation for Sandusky is 34.5 inches per year, with a monthly average of more than 3 inches per month falling in April through September and less than 3 inches in each of the other seven months (The Weather Channel, 2004). Precipitation is fairly evenly distributed throughout the year, with the fewest precipitation days (0.01 inch or greater) per month (10) occurring during July, August, September, and October, and the most

(15) occurring in December and January (City-Data.com, 2004). The mean annual wind speed is 10.3 miles per hour (City-Data.com, 2004), with winds predominantly from the southwest (Science Applications International Corporation, 1991). Sandusky area winters are cloudy with 33 percent sunshine during November through February, as compared with to 65 percent sunshine during the summer months (City-Data.com, 2004).

Geology. Three formations, all of Devonian Age, outcrop across PBOW, each of which was encountered in the upper 100 feet of bedrock at PBOW (Shaw, 2005b). The Delaware Limestone is the lowermost formation screened by site wells. It is characterized as a hard, dense, finely crystalline limestone and dolomite. The unit is typically buff colored and usually is described as fossiliferous. In the vicinity of PBOW, quarries mine limestone from the Delaware. Traces of natural petroleum-derived hydrocarbons and hydrogen sulfide are common in area quarries (Shaw, 2005b). During bedrock well installation and development at WWTP1, hydrogen sulfide was detected in appreciable amounts.

Overlying the Delaware Limestone is the Olentangy Shale. Two members of the Olentangy Shale have been characterized at the site, the Plum Brook Shale and the overlying Prout Limestone. The Plum Brook Shale is interpreted to consist of approximately 35 feet of bluish-gray, soft, fossiliferous shale containing thin layers of dark, hard, fossiliferous limestone. The Prout Limestone has been described as a 15-foot-thick unit which occasionally outcrops in a 1,000-to-2,000-foot-wide, northeast-striking band across the middle portion of PBOW. It is described as a dark-gray to blue, very hard, siliceous, fossiliferous limestone or dolomitic mudstone. The uppermost formation at the site is the Ohio Shale. Only one member of the Ohio Shale is present in the PBOW area, the Huron Shale. This unit has been described as black, thinly bedded, with abundant carbonaceous matter. Some large pyrite/carbonate concretions are also present in the Huron Shale, some as large as 6 feet in diameter (D&M, 1997). Based on drilling logs, the total thickness of shale in the WWTP1/AP1 area is approximately 15 to 19 feet. Shale is not present in the WWTP3/AP3 area.

Soils. The bedrock overburden in Erie County is predominantly glacial till, glacial outwash, or glacial lacustrine (lake) deposits. In the vicinity of PBOW, the soil has been interpreted to be lacustrine. In many areas, the overburden also consists of highly weathered bedrock. The thickness of the overburden ranges from less than 1 foot to greater than 25 feet. Overburden is thickest on the northern portion of the site in the vicinity of the Reactor Facility Area, where it has filled in a bedrock low (Shaw, 2005b). In the WWTP1/AP1 and WWTP3 areas, the overburden thickness ranges from 21 to 26 feet, based on soil boring logs.

The soil in the northwest portion of PBOW is placed within the Kibbie-Elnora-Tuscola-Colwood Association, which is described as nearly level to gently sloping. This soil is described as somewhat poorly drained, moderately well drained, and very poorly drained soils formed in outwash, lacustrine, and deltaic sediments. Along a strip from west to northeast across the site is the Castalia-Millsdale-Milton-Ritchey Association. This association is described as shallow to moderately deep, nearly level to moderately steep, well-drained and very poorly drained soils formed in glacial till, lacustrine sediments, and limestone residuum. Across much of the central portion of the site is the Hornell-Fries-Colwood Association, described as moderately deep to deep, nearly level to gently sloping, somewhat poorly drained to very poorly drained soils formed in glacial till and lacustrine sediments over shale bedrock. At the extreme southeast portion of PBOW is the Pewamo-Bennington Association, described as nearly level to gently sloping, very poorly drained and somewhat poorly drained soils formed from glacial till and lacustrine sediments.

Hydrology. The two main water-bearing zones at PBOW are located in the overburden/shale unit and the limestone bedrock, and are thus called the overburden/shale and bedrock water-bearing zones. The overburden and shale groundwater units show similar water levels in these two units, suggesting good vertical communication. Therefore, these two geologic units were combined for purposes of groundwater evaluation. Data collected during the more recent investigations (Shaw, 2005b; IT, 2001b; 1999; 1997) indicate that groundwater in the overburden is in discontinuous pockets during dry time periods. The shallow overburden generally has low yields over most of the site due to the high percentage of silt and clay. In contrast, the limestone bedrock water-bearing zone is saturated year round. During periods of low precipitation, only limited migration of contaminants would occur in the overburden due to less infiltration. During a wet period, the general flow direction in the overburden water-bearing zone is to the north in the immediate vicinity of the WWTP1/AP1 and WWTP3/AP3. A hydrogeological study by the U.S. Geological Survey (1992) conducted in the glacial deposits of Sandusky in 1990 reported a horizontal hydraulic conductivity of 0.046 feet per day and a vertical hydraulic conductivity of 1.2 feet per day.

Regional groundwater flow in both the overburden/shale and the limestone bedrock is to the north-northeast towards Lake Erie, although local flow may vary due to local topography. Water in the limestone typically occurs in joints and along bedding planes or in solutionally enlarged openings. The conceptual model interprets that bedrock groundwater flow in the Delaware Limestone water-bearing zone is influenced by the frequency, orientation, density, and

connectivity of the bedrock fractures. These fractures result in localized groundwater flow direction to the northwest at WWTP1/AP1 area and to the east-southeast at WWTP3/AP3 area.

At PBOW, the bedrock groundwater has been subdivided into three zones based on location and yield. Zone 1 occurs in the north and northwestern portion of PBOW. It has been characterized as yielding from 100 to 500 gallons per minute (gpm) from karstic limestone approximately 100 feet below grade. Zone 2 is in the northern portion of PBOW and has yields of 15 gpm or less from limestone approximately 300 feet below grade. Zone 3 is located in the eastern and southern portion of the site in predominantly shale bedrock. In addition to being found in the shale, groundwater is located in thin sand and gravel horizons interbedded with silt and clay deposits. Most Zone 3 wells are poor yielding, many of them providing less than 3 gpm (D&M, 1997). WWTP1, AP1, WWTP3, and AP3 are likely in Zone 2.

Surface Water. Surface water features at these WWTPs and APs are generally either ephemeral or consist of drainage ditches adjacent to the sites. No surface water bodies are associated with WWTP1, with the exception of a drainage ditch approximately 300 feet west and northwest of the site. At WWTP3, a drainage ditch is located approximately 30 feet to the west of the former Neutral Waste Storage Tank near the western edge of the site. This drainage ditch connects north of WWTP3 to a larger ditch approximately 50 feet south of Maintenance Area Road. During site reconnaissance visits in October 2008 and April 2009, both of these ditches were water filled but exhibited no noticeable flow.

Two drainage ditches are also present near AP1. A large ditch approximately 10 feet wide and 7 feet deep is present just west of AP1. A culvert from this ditch passes under Maintenance Area Road; during precipitation events, flow would be south-to-north through this culvert. A culvert is also located immediately north of AP1 along Maintenance Road. This culvert accepts runoff from Maintenance Road, may also receive some runoff from AP1, and flows under Maintenance Road into a ditch. This small ditch, approximately 5 feet wide and 3 feet deep, branches off the main ditch and drains to the larger ditch (described above) north of Maintenance Road.

AP3 had formerly been filled with water during former PBOW Power House 3 and NASA K-Site operations. Water is still present at times in AP3. A creek bed extends from AP3 to the west but was dry during the site reconnaissance.

Additional descriptions of the individual sites are provided in the following subsections.

3.1.1.1 WWTP1

The former WWTP1 is located in the northern midsection of PBOW, approximately 200 feet northeast of the intersection of Maintenance Road and Taylor Road (Figure 1-2). It occupies approximately 2 acres of land. A former building located approximately 200 feet northeast of WWTP1 is still present and is currently used by NASA.

Limited investigation has been conducted at WWTP1. Historical drawings were reviewed by D&M (1997), and it was determined that the raw waste storage tank located at the former WWTP1 received waste from the settling basins located at manufacturing areas TNTA and TNTB.

During a field reconnaissance in March 1999, bare areas and the remains of concrete foundations were reported to have been observed where the former buildings and storage tanks were once present (USACE, 2000a). During field reconnaissance in October 2008 and April 2009 by Shaw, open, grass-covered areas were present at the location of former buildings, but few of the foundations were visible, either having been removed or grown over by vegetation. A few inches of water was observed in the drainage ditch west and northwest of WWTP1, but no flow was observed.

3.1.1.2 WWTP3

Former WWTP3 is located in the western midsection of PBOW, approximately 650 feet southwest of the intersection of Maintenance Road and Ransom Road and 450 feet southeast of the patrol road. It occupies approximately 2 acres of land (Figure 1-2). The neutralized waste water storage tank of former WWTP3 is had been used by NASA as the K-Site control building (USACE, 2000a). The K-Site itself was operated in the former Power House 3 building. D&M (1995) conducted a review of records and found that a bermed, square depression north of WWTP3 and south of Maintenance Road may have been used as an ash settling basin. Water was observed during field reconnaissance in portions of the drainage ditch located west of WWTP3 in both October 2008 and April 2009, but no flow was observed.

3.1.1.3 AP1

Former AP1 is located in the central part of PBOW, approximately 1,095 feet west of the intersection of Maintenance Road and Taylor Road and approximately 50 feet south of Maintenance Road (Figure 1-2).

In June 1999, USACE conducted a limited site investigation of AP1 (USACE, 2000b). AP1 was noted to be overgrown with thick vegetation and a thicket of small trees ranging from 3 to 8 feet tall. During a field reconnaissance by Shaw and USACE personnel in April 2009, mostly grey dogwood (*Cornus racemosa*) trees were present in AP1 at heights of approximately 15 feet. Personnel were informed by NASA that an underground, high-voltage utility line bisected AP1 in a north-south direction and an underground telephone line was present approximately 5 feet south and parallel to Maintenance Road. The USACE also noted a culvert under Maintenance Road and an associated drainage ditch approximately 5 feet due north of AP1. A larger drainage ditch located northwest of AP1 contained water that reportedly flowed in a northeast direction. During October 2008 and April 2009 site visits, USACE and Shaw personnel observed water in this ditch, but it did not appear to be flowing. Standing water was observed in the far north portion of AP1 during April 2009 reconnaissance, especially in ruts made by vehicles and other equipment along Maintenance Road. This standing water was not observed during site visits in September or October 2008. A culvert receives runoff from Maintenance Road and possibly from AP1, and drains to the north under Maintenance Road.

3.1.1.4 AP3

AP3 is located approximately 800 feet southwest of the intersection of Maintenance and Ransom Road and is west of the former NASA K-Site Test Facility research building (former Power House 3) (Figure 1-2). Operations at the K-Site were officially abandoned in 2007; however, it is possible that the facility may be used for test programs in the future. Abandoned railroad tracks running in a north-south direction are immediately east of AP3. The pit is partially surrounded by thick vegetation, with mature and smaller trees. The original surface water impoundment was observed holding water supplied by NASA's K-Site Test Facility in 1999 (USACE, 2000b). Water had been discharged from the former pond by means of an east-west-trending drainage ditch that eventually discharges into Pipe Creek (USACE, 2000b).
Since the K-Site ceased operations, the major source of water to Ash Pit 3 has apparently been eliminated. A review of aerial photographs indicates a lack of open water at AP3 in recent history. NASA (2008) personnel stated that no water was present in AP3 on July 22, 2008. This was also the case during a September 2008 site visit by USACE and Shaw personnel, although a small ponded area was observed after a rain event during an October 2008 site visit. However, much of AP3 was covered by up to approximately 1 foot of water during an April 2009 reconnaissance. This 2009 reconnaissance took place after a precipitation event during a wetter than usual spring.

3.1.2 Contaminant Sources, Release Mechanisms, and Migration Pathways

Contaminant sources, release mechanisms, and migration pathways are summarized on Figure 3-1. Briefly, TNT is made by nitrating toluene in a three-step process that uses nitric and sulfuric acids (D&M, 1997). The waste water treatment plants received waste water from the TNT operations, and discharged the treated liquid to open ponds. The thickened liquid was incinerated, and the ash from this incineration was disposed of in nearby ash pits. Contamination resulted from the inadvertent release of TNT, its precursors, contaminants and residues, and acids or sellite (sodium sulfite made from soda ash and sulfur) from the process lines or drying areas. Releases occurred to the surface soil and to subsurface soil from vertical migration. Runoff and erosion may have spread contamination over the surrounding surface soil and may have carried contaminants to nearby streams. Infiltration and leaching may have carried contaminants into the subsurface soil or groundwater.

3.1.3 Receptors and Exposure Pathways

Receptors, selected to represent the upper bound on exposure from all plausibly exposed groups of people at WWTPs 1 and 3 and APs 1 and 3, as well as the pathways by which they may be exposed to chemicals, are summarized On Figure 3-1 and in Table 3-1. The exposure variable values used in the contaminant intake models are compiled in Table 3-2. The receptors to be evaluated in the human health risk assessment include the following:

- Current and future groundskeeper
- Current and future construction worker
- Future on-site resident
- Future indoor worker
- Future hunter
- Future hunter's child.

Most BHHRAs are based on a reasonable maximum exposure (RME) assumption. The intent of the RME assumption is to estimate the highest exposure level that could reasonably be expected to occur, but not necessarily the worst possible case (EPA, 1989a; 1991a). It is interpreted as reflecting the 90 to 95th percentile on exposure. In keeping with EPA (1989a; 1991a) guidance, variables chosen for a baseline RME scenario for ingestion rate, exposure frequency, and exposure duration are generally upper bounds. Other variables, such as body weight and exposed skin surface area, are generally central or average values. In the case of contact rates consisting of multiple components, e.g., dermal contact with soil or water, which consists of a dermal absorption factor (ABS) and soil-to-skin adherence factor (AF) for soil, and permeability coefficient (K_p) and exposure time for water, only one variable, ABS or K_p, needs to be an upper

bound. The conservativeness built into the individual variables ensures that the entire estimate for contact rate is sufficiently conservative.

The averaging time for noncancer evaluation is computed as the product of the exposure duration (years) multiplied by 365 days per year. The resultant noncancer averaging time is used to estimate an average daily dose over the entire exposure period (EPA, 1989a). For cancer evaluation, the averaging time is computed as the product of 70 years, the assumed human lifetime, times 365 days/year. This cancer-based averaging time is used to estimate an average daily dose prorated over a lifetime, regardless of the frequency or duration of exposure. The methodology used in deriving the averaging time for cancer risks assumes that the risk from short-term exposure to a high dose of a given carcinogen is equivalent to long-term exposure to a correspondingly lower dose, provided that the total lifetime doses are equivalent. This approach is generally consistent with the EPA (1986) policy of carcinogen evaluation, although it introduces considerable uncertainty into the BHHRA cancer risk estimates.

A fractional term (FI) is introduced into the chemical intake equations to account for scenarios in which exposure to a potentially contaminated medium associated with the site is less than total daily exposure to that medium. For example, if the site of interest is small, so that a groundskeeper may spend only one-half of his working time at the site, an FI of 0.5 might be applied to the soil ingestion and dermal intake equations. An FI is used also if a receptor's exposure is split between two comparable media. For example, if a resident is exposed to both soil and sediment, FIs may be introduced that apportion his exposure between the two media such that the FI value for the two analogous media does not exceed a value of 1 (see Section 3.1.3.4).

3.1.3.1 Groundskeeper

The groundskeeper scenario is designed to evaluate the upper bound for long-term site worker exposure to surface soil in the current site-use scenario and total soil in the future site-use scenario. Total soil is evaluated under the future-use scenario as hypothetical future construction may include considerable excavation of subsurface soil. This soil may be spread on the surface and regraded such that some of the soil currently in the subsurface (i.e., 1-10 ft bgs) will be spread as surface soil (0-1 ft bgs). Groundwater use is also evaluated for the groundskeeper in the future site-use scenario, as discussed in Section 3.1.

Direct soil exposure pathways include incidental ingestion and dermal contact. Inhalation of dust, raised by operating lawn mowers or other equipment, is also evaluated because relatively

high dust concentrations may be produced within the groundskeeper's breathing zone, with little opportunity for dilution by the large volume of ambient air.

Shaw's experience has been that surface soil contaminated with volatile organic compounds (VOC) and that has been in place for extended periods is not a significant source of airborne VOCs, because infiltration and dissipation over time reduces residues at the surface (i.e., first few centimeters) from which volatilization would occur. However, as noted above, the data set for surface soil may include samples taken from up to 1 ft bgs, which would include the soil zone deeper than the top few centimeters, where dissipation has not reduced VOC concentrations. In other words, the surface soil data set might indicate the presence of VOCs, although volatilization to the air is unlikely to be significant. Therefore, a surface soil-to-air volatilization model will not be used in addition to the activity-based dust emissions model to estimate airborne concentrations of VOCs. Instead, the airborne concentrations estimated by the dust emissions model will be assumed to sufficiently estimate levels of VOCs that may arise from volatilization, because the dust emissions model treats the VOCs as if they were located at the surface. It is assumed that VOC emissions from subsurface soil (i.e., at depths greater than 1 ft bgs) would be attenuated by the overlying soil so that concentrations in ambient air would not be toxicologically significant.

It is assumed that any contact with surface water or sediment in the ditches associated with WWTPs 1 and 3 and APs 1 and 3 would be infrequent and sporadic, because such contact would not be a part of the groundskeeper's regular duties or activities. Therefore, exposure to these media is not quantified. Under a future land-use scenario, a groundskeeper may be exposed to groundwater, which could theoretically be developed as a source of drinking water.

The groundskeeper is assumed to be a 70-kilogram (kg) adult who works 8 hours per day, approximately 5 days per week year-round on site for a total of 250 days/year for 25 years (EPA, 2004a). The respiratory rate for the groundskeeper is assumed to be 20 cubic meters (m^3) per 8-hour workday or 2.5 cubic meters per hour (m^3/hr) (EPA, 1991a), and the soil incidental ingestion rate is assumed to be 100 milligrams per day (mg/day) (EPA, 2002). The groundskeeper is assumed to be exposed dermally to soil. An exposed skin surface area of 3,300 square centimeters (cm^2) and a soil AF of 0.2 milligrams per cm^2 (mg/cm^2) are assumed (EPA, 2004a).

A future-scenario groundskeeper is assumed to be exposed to groundwater, which could theoretically be developed as a source of drinking water at sometime in the future. His drinking

water ingestion rate is assumed to be 1 liter per day (L/day) (EPA, 1991a). He may also experience dermal contact with groundwater used to clean equipment and to rinse dust or perspiration from his body. For this evaluation, it is assumed that the head, forearms, and hands, approximately 3,300 cm^2 of his body (EPA, 2004a), would be exposed intermittently for up to 1 hour/day. Because exposure is assumed to be intermittent rather than continuous, organic chemical uptake across the dermis would not reach steady state, which guides the selection of the EPA (2004a) model used to quantify this pathway (see Section 3.3.4).

3.1.3.2 Indoor Worker

This receptor scenario was created to evaluate exposure to indoor airborne VOCs entrapped in a building. VOCs released from subsurface soil may enter a building through joints or cracks in the foundation or slab. The indoor worker is also potentially exposed to surface soil via incidental ingestion. Exposure to COPCs in surface soil via dermal contact and inhalation of airborne dust and VOCs from surface soil, although plausible, are expected to be less significant than incidental ingestion, because this receptor spends his work time indoors. Therefore, dermal contact and inhalation of dust and airborne VOCs from surface soil are not quantified separately from ingestion exposure (EPA, 2002). Under a future-use scenario for this receptor, construction of a building would be necessary. This would require excavation and regrading of soil. Normally, when construction is involved, such as for the future groundskeeper or resident, total soil rather than surface soil would be evaluated for ingestion exposure. However, the chief purpose for this receptor is to evaluate exposure via vapor intrusion of contaminants from subsurface soil into indoor air. Thus, the evaluation of direct contact with subsurface soil as a component of total soil would equate to "double counting" of COPCs in subsurface soil. Also, the groundskeeper reflects a worst-case exposure for a long-term worker with respect to direct contact with both surface soil and total soil. Therefore, the direct contact component for the indoor worker is included to reflect a more complete exposure scenario, but direct-contact soil exposure is most effectively addressed from an RME perspective by the groundskeeper.

The indoor worker is assumed to be a 70-kg adult who works 8 hours/day, approximately 5 days/week year-round on the site for a total of 250 days/year for 25 years (EPA, 2002). His incidental soil ingestion rate is assumed to be 50 mg/day (EPA, 2002), and his inhalation rate is assumed to be 20 m^3/8-hour workday (EPA, 1991a).

A future indoor worker is assumed to be exposed to groundwater, which could theoretically be developed as a source of drinking water (see Section 3.1). His drinking water ingestion rate is assumed to be 1L/day (EPA, 1991a). Some indoor worker positions may require relatively

frequent dermal contact with groundwater as well, e.g., a food preparer/cafeteria worker that would wash his hands, produce, equipment, etc. For this evaluation, it is assumed that the head, forearms, and hands, approximately 3,300 cm^2 of his body (EPA, 2004a), would be exposed intermittently for up to 1 hour per day. Because exposure is assumed to be intermittent rather than continuous, organic chemical uptake across the dermis would not reach steady state, which guides the selection of the EPA (2004a) model used to quantify this pathway (see Section 3.3.4).

3.1.3.3 Construction Worker

The construction worker scenario is used to evaluate short-term exposure to surface and subsurface soil (total soil) in either the current or future land-use scenario. Construction projects are expected to be infrequent. It is assumed that the construction worker participates in only one construction project on the site. Relevant exposure pathways include incidental ingestion and dermal contact, inhalation of dust raised by operating construction equipment, and inhalation of airborne VOCs released from subsurface soil during excavation and grading. Exposure to groundwater by the construction worker is also possible; however, if on-site groundwater were developed as a tap water source, other potential future groundwater receptors such as the groundskeeper would have longer and/or more frequent groundwater exposure. Therefore, groundwater exposure is not evaluated for the construction worker.

The construction worker may also be exposed to surface water and sediment during projects such as installation of underground utilities or re-routing stream flow. Dermal contact is the most significant pathway for exposure to surface water. Incidental ingestion of surface water is also possible, but is not expected to be nearly as significant as dermal contact. Inhalation of VOCs from surface water is also possible, but the large volume of outdoor air and natural air currents are expected to dilute airborne concentrations, so that this pathway is expected to be less significant than dermal contact, which is quantified. For these reasons, incidental ingestion and inhalation of VOCs from surface water are not quantified separately from dermal contact. Dermal contact and incidental ingestion may be important pathways for exposure to sediment, and both are evaluated.

The construction worker is assumed to be a 70-kg adult who works 8 hours/day, approximately 5 days/week (EPA, 2002). Construction projects involving soil exposure are assumed to last 6 months. The respiratory rate for the construction worker is assumed to be 20 m^3/8-hour workday (2.5 m^3/hour) (EPA, 1991a). A soil ingestion rate of 330 mg/day is assumed for the construction worker (EPA, 2002). A dermal soil AF for the construction worker of 0.3 mg/cm^2 and an

exposed body surface area of 3,300 cm^2 are assumed, which represent the head, hands, and forearms (EPA, 2002).

The construction worker may be exposed to surface water and sediment during the 6-month construction period. The construction worker dermal exposure parameters for sediment are assumed to be exactly the same as those for soil. Dermal exposure to surface water is assumed to occur for up to 4 hours per day or one-half the normal work day. It is assumed the exposure to surface water is intermittent during this period. An exposed body surface area of 3,300 cm^2, the same as for sediment and soil, is assumed for exposure of the construction worker to surface water. It is expected that the construction worker would wear appropriate footgear and leg protection to minimize surface water and sediment exposure to the legs.

The construction worker scenario described above provides for several different kinds of construction projects, such as upland excavation and building projects (exposure primarily to soil), as well as stream re-routing (exposure primarily to surface water and sediment). It is unlikely, however, that a single construction worker would participate in all these activities during a given project. Therefore, the evaluation described above is probably overly conservative and may represent some double counting. For example, it is unlikely that the construction worker would be simultaneously ingesting soil, sediment, and surface water. Similarly, the air in his breathing zone is not likely to contain the reasonable maximum concentrations of COPCs estimated for soil while he is exposed to surface water. The potential for double-counting is not expected to contribute significantly to total risk estimates summed across chemicals, pathways and media. Should construction worker risk estimates exceed acceptable limits, risk and hazard estimates may be performed using refined exposure assumptions based on the physical characteristics of the site. For example, an upland excavation and building project may be assumed for one or more areas of the site, and a stream re-routing project may be assumed for another. Effectively, the risks and hazards associated with surface water/sediment exposure and soil exposure could be separated. This approach would more precisely reflect plausible exposure scenarios, reduce the likelihood of double counting, and more accurately identify risk-driving media and chemicals. These refined estimates would be presented in the uncertainty analysis of the BHHRA.

3.1.3.4 On-Site Resident

The on-site residential scenario is created to evaluate the upper bound for long-term exposure to site soil, surface water, sediment, and groundwater under the future land-use scenario. The on-site residential scenario is evaluated assuming a 30-residential exposure scenario, considering

exposure to a resident as a young child (6-year duration, ages 1 through 6 years) through adult portion of life spent at this residence (24-year duration) (EPA, 1991a).. Noncancer hazard estimates will be derived separately for the child and adult life stages. Cancer risk is estimated as the sum of the risks calculated for the adult (24 years) and the child (6 years) (EPA, 2002; 2004c).

The resident is assumed to be exposed directly to total soil, because residential development would involve excavation and regrading, which would mix surface and subsurface soil. Relevant pathways for total soil exposure include incidental ingestion, dermal contact, and inhalation of dust and VOCs. Evaluation of VOCs from total soil is addressed during evaluation of airborne dust, as described for the groundskeeper. For evaluating inhalation of airborne dust, it is assumed that 80 percent of the soil surface is covered with pavement or vegetation. Inhalation of VOCs released from subsurface soil entrapped in indoor air is also evaluated. The resident is also assumed to be exposed to VOCs that have been released from subsurface soil through cracks in the building foundation to indoor air. It is noted that because some of the subsurface soil is expected to be brought to the surface in the future, using only subsurface soil data will conservatively result in some double counting of exposure to VOCs in the subsurface soil. This will be addressed in the uncertainty analysis if the subsurface soil-to-indoor air pathway significantly affects risk and hazard estimates.

It will be assumed that, if appropriate based on the RI data, under future residential land use the overburden and limestone bedrock water units will be developed as sources of potable water (see Section 3.1). The resident will be assumed to use groundwater underlying the site as the sole source of household tap water. Exposure to COPCs in groundwater would occur via ingestion, dermal contact during bathing/washing, and inhalation of VOCs released to the air during household use of tap water associated with multiple household uses.

The resident could have access to the ditches associated with WWTPs 1 and 3 and APs 1 and 3, and could be exposed to contaminants in surface water and sediment. Plausible exposure pathways include dermal contact with surface water and incidental ingestion and dermal contact with sediment. Incidental ingestion of surface water in a wading scenario is considered less significant than dermal contact and is not quantified separately from dermal contact. Inhalation of VOC emissions from surface water is also possible, but the large volume of outdoor air and natural air currents are expected to dilute airborne concentrations, so that this pathway is expected to be less significant than dermal contact, which is quantified. For these reasons, the

inhalation of VOC emissions from surface water is not quantified separately from dermal contact.

The adult resident is assumed to be a 70-kg person with an incidental soil ingestion rate of 100 mg/day and an inhalation rate of 20 cubic meters per day (m^3/day) or 0.83m^3/hr (EPA, 1991a). A body surface area of 5,700 cm^2, representing the hands, forearms, head and lower legs will be assumed to be available for dermal exposure to soil (EPA, 2004a). A soil AF of 0.07 mg/cm^2 is used as the default RME value for the adult resident (EPA, 2004a). The adult resident is assumed to be exposed for 350 days/year for 24 years (EPA, 1991a; 2002).

The child resident is assumed to be a 1- through 6-year-old child with an average body weight of 15 kg, a soil ingestion rate of 200 mg/day, and an inhalation rate of 10 m^3/day or 0.417 m^3/hr (EPA, 2004c). A body surface area of 2,800 cm^2, representing the head, hands, forearms, lower legs, and feet, is assumed for dermal contact with soil (EPA, 2004a). A soil AF of 0.2 mg/cm^2 is used as the default RME value for the child resident (EPA, 2004a). The child resident is assumed to be exposed for 350 days/year for 6 years (EPA, 1991a; 2002).

It is assumed that the resident would visit the ditches for 8 hours/day, 2 days/week during the warmer half of the year. The resident is assumed to wade for 3 hours/day on 52 days of the year. Mechanisms of exposure to soil and sediment are likely to be similar. Therefore, the incidental soil ingestion rate of 100 mg/day, the surface area of 5,700 cm^2, and the AF of 0.07 mg/cm^2 are also applied to sediment exposure in the adult. Similarly, the resident child soil ingestion rate of 200 mg/day, skin surface area of 2,800 cm^2, and soil AF of 0.2 mg/cm^2 will be applied to sediment exposure for this receptor. The ditches generally contain only a few inches of nonflowing water, limiting the surface area of the body that would typically be exposed. It will be assumed that an adult body surface area of 7,000 cm^2 will be available for exposure to surface water. This represents the same body parts to which soil and sediment would be exposed (i.e., hands, forearms, head, and lower legs) plus the feet (EPA, 2004a; 1997a). The body surface area of 2,800 cm^2, representing the hands, forearms, head, lower legs, and feet, used for soil and sediment exposure in the young child will also be used for surface water exposure for this receptor.

EPA (1989a) permits the development of an FI to reflect the proportion of total daily exposure that a receptor obtains from potentially contaminated medium (refer to Section 3.1.3). For this receptor, the FI is used to apportion the resident's time of exposure between site soil and sediment. It is assumed that the resident spends 16 hours/day awake and potentially exposed to

soil or sediment. As previously noted, 350 days/year are available for contact with soil; 52 of those days are also available for contact with sediment. It is assumed that contact with soil and sediment does not occur simultaneously; i.e., on those days when the resident spends time at the streams, 8 hours would be spent in contact with soil and 8 hours would be spent in contact with sediment. Therefore, the fraction of exposure to soil is 16 hours/16 hours = 1 on the 298 days without time spent at the streams, and the fraction of exposure to soil is estimated as 8 hours/16 hours = 0.5 on the 52 days with some time spent at the streams. A weighted fraction of 0.93 (rounded to 0.9) is estimated for exposure to soil over the entire 350 days/year exposure frequency. A weighted fraction of 0.07 (rounded to 0.1) is estimated for exposure to sediment over the entire 350 days/year exposure frequency.

An adolescent resident may be the most likely individual to have regular exposure with sediment and surface water associated with these ditches. It is not expected that adults would regularly visit these ditches, as these ditches do not support game fish and would seemingly not provide any attraction. It is unlikely that a young child (i.e., ages 1 through 6) would frequent these ditches for substantial portions of time, because such young children (especially at the lower end of this age range) would require continued adult supervision. However, it will be conservatively assumed, as described above, that the resident will be regularly exposed to surface water and sediment for 30 years, 6 years assumed as a young child and 24 years as an adult. Note that for noncancer effects, only the young child resident receptor will be evaluated, as young children receive a proportionately greater level of exposure than do adults. For cancer effects, the 30-year exposure to surface water and sediment represented by both the young child and adult will be combined. This approach is more conservative than evaluating an adolescent and is also consistent with BHHRAs performed for PBOW sites in the past. If appropriate, the conservativeness of these assumptions will be evaluated in the uncertainty analysis of the BHHRA.

With respect to groundwater exposure, it is assumed that an adult resident ingests 2 L/day of tap water (EPA, 1991a), and that the young child drinks 1 L/day (EPA, 2004c). The total body surface areas of the adult and of the young child resident are assumed to be exposed to tap water while bathing/showering. The total surface area for an adult is assumed to be 18,000 cm^2 and the total surface area for the young child is assumed to be 6,600 cm^2. Both the child and adult resident are assumed to be dermally exposed to COPCs in groundwater while bathing/showering. The child will be assumed to bathe for 1 hour per day (EPA, 2004a), and the adult will be assumed to shower for 35 minutes per day (0.58 hour/day) (EPA, 2004a). Inhalation rates of 0.833 m^3/hr for the adult (EPA, 1991a) and 0.416 m^3/hr for the child (EPA, 2004c) will be used.

Because EPA, (1997a) lists a 90th percentile for time spent in a residence as over 23 hours per day, it will be conservatively assumed that the resident spends 24 hours per day in the house.

3.1.3.5 Hunter

This scenario is created to evaluate the potential for contaminants in soil to affect food chain pathways. Both WWTPs and APs provide habitat for deer and other wildlife, and deer hunting is permitted on the PBOW facility. Although hunting is not currently permitted in any of these WWTP or AP sites, it may be permitted in these areas in the future. Therefore, a hunter who consumes his game is a plausible future scenario requiring evaluation. Many kinds of game animals may be hunted and consumed (e.g., squirrel, pheasant and other upland birds, turkey, deer); however, the deer is the species most likely to contribute meaningfully to the diet. Therefore, this evaluation is limited to a deer hunter. Potential exposure pathways include incidental soil ingestion, dermal contact with soil, and ingestion of venison from deer that browse plants growing on contaminated surface soil, all of which are evaluated quantitatively. Inhalation of airborne dust from wind currents is a potentially complete exposure pathway; however, vegetation reduces dust emissions to insignificant levels (EPA, 1996), and it is assumed that the deer hunter would spend virtually all of his time on vegetated rather than bare soil. Therefore, it is assumed that inhalation exposure would contribute much less than incidental ingestion, and the inhalation exposure pathway is not quantified separately from ingestion.

Inhalation exposure to airborne VOCs from subsurface soil and surface water is not evaluated for the reasons previously explained for other receptors. Although exposure to surface water and sediment is possible, it would likely be short-term and sporadic because such contact would not be part of the hunter's regular activities. Therefore, exposure to these media will not be quantified.

The deer hunter is assumed to be a 70-kg adult who harvests deer and consumes venison over a 30-year period. It is assumed that he spends 14 days per year hunting on PBOW. His incidental soil ingestion rate is assumed to be 100 mg/day (EPA, 1991a). Hunting at PBOW occurs in the fall and winter. Given the temperate climate of northern Ohio during hunting season, a hunter would dress appropriately, with typically only the hands and head exposed, at most. The default industrial RME exposed skin surface area of 3,300 cm^2, which represents the hands, forearms, and head (EPA, 2004a), will be conservatively assumed for the hunter. The default industrial RME soil AF of 0.2 mg/cm^2 (EPA, 2004a) will also be assumed.

Data were not located regarding the rate of venison ingestion; therefore, a hypothetical scenario is adapted from the assumptions applied to a similar site in West Virginia (IT, 2000) and subsequently applied to TNTA and TNTC (IT, 2001a). A highly conservative but plausible scenario consists of a hunter who kills a deer each year from the WWTP or AP site under evaluation. It is assumed that the hunter eats 10 pounds (4.5 kg) of venison per year (Sharp, 1995). This consumption rate corresponds to 0.013 kilograms per day (kg/day) (0.186 grams per kilogram of body weight per day [g/kg-day]) of venison for each of the 350 days per year (EPA, 1991a) that the hunter spends at his residence.

3.1.3.6 Hunter's Child

It is likely that a successful hunter, described in Section 3.1.3.5, would share his venison with the rest of the family, which may include small children. Small children, however, would be unlikely to accompany the hunter afield. Therefore, the direct exposure pathways evaluated for the hunter (i.e., incidental ingestion and dermal contact with soil) will not be evaluated for the small child.

Data regarding the rate of venison ingestion by small children were not located. However, if it is assumed that venison may replace beef in the diet, the differences in beef consumption between adults and children can be used to estimate a venison ingestion rate for children. EPA (1997a) provides per capita beef intake data for <1- to 5-year-old children ranging from 0.941 to 1.46 g/kg-day (time-weighted average of 1.296 g/kg-day). EPA (1997a) provides per capita beef intake data for 12- to 70+-year-old adults ranging from 0.568 to 0.83 g/kg-day (time-weighted average of 0.727 g/kg-day). From these data, it can be estimated that the rate of beef consumption for small children, expressed on a body weight basis, is approximately 1.8 times that of an adult. Therefore, a venison ingestion rate of 0.335 g/kg-day is estimated for a young child from the venison ingestion rate of 0.186 g/kg-day for the adult. Assuming that the child is 1 through 6 years old with an average body weight of 15 kg (EPA, 1991a; 2002), the child's venison ingestion rate may be expressed as 0.005 kg/day.

3.1.3.7 Other Receptors Not Considered

Another plausible receptor group is delivery personnel. These receptors, however, would be less intensively exposed to soil than the groundskeeper; therefore, their exposures are not evaluated. WWTPs 1 and 3 and APs 1 and 3 could become part of the area used for National Guard training activities. National Guard trainees, however, may be less exposed to any of the potentially contaminated media than the receptors identified above. Because they would likely not represent an upper bound for nonresidential exposure, these receptors are not evaluated. Parts of PBOW

are used for fishing and hunting. The streams near WWTPs 1 and 3 and APs 1 and 3, however, are too small to support game fish. Therefore, fish ingestion as an indirect pathway for exposure to surface water and sediment is not evaluated.

Another potential receptor is an off-site resident. It is assumed, however, that the higher concentrations of contaminants occur on site; therefore, the on-site resident would be the more heavily exposed, and the off-site resident is not evaluated.

3.2 Quantification of Exposure-Point Concentrations

The EPC is an estimate of the concentration of a COPC in a given medium to which a receptor may be exposed over the duration of the exposure. An EPC may be based on media concentrations that have been directly measured using laboratory analysis, or it may be derived based on environmental medium-to-medium transport modeling. The EPCs of COPCs in soil, groundwater, surface water, and sediment will be derived based on measured analytical data. Note that the EPC for dermal exposure to VOCs in groundwater is based on one-half the EPC concentration derived from the measured concentrations in groundwater (Sections 3.2.1 and 3.2.2.5). This value is used because it is assumed that 50 percent of the groundwater VOC concentration will be volatilized during normal household use (Section 3.2.2.4). Concentrations of COPCs in air and venison will not be measured (and in some cases cannot reasonably be measured) but will be based on models that use the EPCs of COPCs in the appropriate directly-measured media (i.e., soil and groundwater) as input values.

Section 3.2.1 describes the approaches used to derive EPCs for direct exposure to soil, groundwater, surface water, and sediment based on analytical measurements from samples of these media. Models to derive EPCs for the air are described in Sections 3.2.2.1 through 3.2.2.4, and the model used to derive venison EPCs is described in Section 3.2.2.6.

3.2.1 Soil, Groundwater, Surface Water, and Sediment Concentrations

Exposure to an environmental medium is generally assumed to be random, and the EPC should be the arithmetic average encountered over the duration of exposure (EPA, 1989a). Therefore, the population mean concentration, if known, would be the ideal value selected as the EPC. The sample mean is an obvious estimate of the population mean. However, uncertainties exist as to how well the sample mean represents the population mean. Therefore, EPA (1989a) has recommended the inclusion of a UCL for RME evaluation as a conservative estimate of the true mean exposure concentration.

The EPA (2009a,b,c) ProUCL (Version 4.00.04) software will be used to estimate UCLs for the data sets of all environmental media represented by at least five samples. If the data set consists of fewer than five data points, the MDC will be selected as the EPC. Analytical data from field duplicates are averaged with originals to yield one result for use in the statistical manipulations (see Section 2.5). The method detection limit will be used as the ProUCL input concentration for nondetects. Nondetects with method detection limits greater than the MDC will not be included in the data set used to calculate the EPC (EPA, 1989), as such values unduly make distribution testing impossible for ProUCL (EPA, 2009c). If any sample results are eliminated based on high method detection limits, these will be identified in the data evaluation and discussed in the uncertainty analysis of the BHHRA.

ProUCL generates a variety of UCL estimates for each data set. Generally, the results of one or two (sometimes more) of the UCL estimates are recommended. This recommendation is based on a variety of factors including the distribution (i.e., normal, lognormal, gamma, or not discernable) that provides the best fit, number of nondetects, size of the data set, and skewness. In general, the UCL recommended by ProUCL will be selected as the EPC. Occasionally, ProUCL will recommend the 97.5 or 99 percent UCL on the arithmetic mean estimated by the Chebyshev method. In these cases, the 95 percent UCL estimated by the Chebyshev method is selected as the EPC because this is more consistent with the intent of the RME paradigm as defined by EPA (1989a; 1991a).

The UCL generated by ProUCL or the MDC, whichever is smaller, will be selected as the EPC and is understood to represent a conservative estimate of average for use in the risk assessment or in various transport models used to estimate EPCs. Unusually high detected values are included in the calculation of the UCL concentration. Inclusion of these high values increases the statistical variability and the overall conservativeness of the risk estimate.

ProUCL is a software tool that provides estimates of the UCL using a variety of mathematical approaches. As mentioned, its output includes one or more recommendations. Depending on the data set, some of the estimates generated by the various calculation methods included in ProUCL may vary by an order of magnitude. ProUCL and the decision tree on which its recommendations are based have been developed using multitudes of simulated data sets with a variety of distributions and other characteristics. There are uncertainties as to how well this decision tree will derive a recommended UCL for a given data set. This uncertainty tends to increase with variability, skewness, and where a large number of the samples are nondetects. For example, with respect to distribution testing, ProUCL bases the determination of distribution

type only on the detected samples. The true concentrations of the nondetected values are unknown, and this lack of information can affect the distribution determination and consequently affect the ProUCL recommendation. The general uncertainties associated with the EPC values and the use of ProUCL will be discussed in the uncertainty analysis of the BHHRA (Chapter 6.0). Specific uncertainties associated with the EPC values of specific data sets will be discussed in the uncertainty analysis as appropriate.

3.2.2 Exposure-Point Concentrations in Air

3.2.2.1 COPC Concentrations from Dust

Inhalation exposure to particulate (dust) emissions from soils for the groundskeeper and construction worker evaluations arises from activities that raise dust. Therefore, the most appropriate approach to estimating chemical concentrations in ambient air is the use an activity-based dust loading equation (U.S. Department of Energy [DOE], 1989):

$$C_a = (D)(C_s)(CF_1) \qquad \text{Eq. 3.1}$$

where:

C_a = contaminant concentration in air (milligrams per cubic meter [mg/m^3], calculated)
D = dust loading factor (g of soil/m^3 of air)
C_s = contaminant concentration in soil (milligrams per kilogram [mg/kg])
CF_1 = conversion factor (1E-3 kilograms per gram).

Plausible values for D include 2E-4 grams per cubic meter (g/m^3) for agricultural activity (DOE, 1989), 6E-4 g/m^3 for construction work (DOE, 1983), and 1E-4 g/m^3 for other activity (National Council on Radiation Protection and Measurements, 1984). The value for D of 1E-4 g/m^3 for other activity is used for the groundskeeper. It is assumed that construction activities requiring intimate contact with soil, for which D = 6E-4 g/m^3 is appropriate, may last for one-half of a construction period. The remaining one-half of the time is more realistically characterized by D = 1E-4 g/m^3. Therefore, a time-weighted average dust loading factor for construction work of 3.5E-4 g/m^3 is estimated for the construction worker.

Airborne concentrations of VOCs estimated by the dust loading model will be assumed to sufficiently estimate levels of VOCs that may arise from volatilization, because the dust loading model treats the VOCs as if they were located at the ground surface.

The resident is more likely to be exposed to dust arising from wind erosion than from dust-raising activities on the site. EPA (1996) derived a model for estimating a dust particulate emission factor based on an "unlimited reservoir" model and the assumption that the source area is square:

Eq. 3.2

$$PEF = Q/C \times \frac{3600}{0.036 \times (1 - V) \times (U_m / U_t)^3 \times F(x)}$$

where:

PEF	=	particulate emission factor (cubic meters per kilogram [m^3/kg], calculated)
Q/C	=	inverse of the mean concentration at center of square source (55.99 grams per square meter [g/m^2]-second per kg/m^3, site-specific value from Table 3 in EPA [1996] [Zone 7, Cleveland, 5-acre site])
3600	=	seconds/hour
V	=	fraction of surface covered with vegetation (0.8, unitless, assumed)
U_m	=	mean annual wind speed (default, 4.60 meters per second [m/second] equals mean annual wind speed of 10.3 miles per hour [see Section 3.1.1])
U_t	=	equivalent threshold value of wind speed at 7 m (default, 11.32 m/second)
F(x)	=	function dependent on U_m/U_t (default, 0.194).

The concentration of COPC in air is calculated as follows:

Eq. 3.3

$$C_a = \frac{C_s}{PEF}$$

where:

C_a	=	contaminant concentration in air (mg/m^3, calculated)
C_s	=	contaminant concentration in soil (mg/kg)
PEF	=	particulate emission factor (m^3/kg).

Airborne concentrations of VOCs estimated by the wind erosion model will be assumed to sufficiently estimate levels of VOCs that may arise from volatilization, because the wind erosion model treats the VOCs as if they were located at the ground surface.

3.2.2.2 COPC Concentrations in Indoor Air

An EPA (2004d) modification of the Johnson and Ettinger (1991) model is used to estimate airborne concentrations of VOCs in indoor air from vapor intrusion associated with contaminants

in subsurface soil for the indoor worker and resident. A typical single-family residential home is assumed for both the resident and future on-site worker. Note that the parameters used to model residential homes are typically more conservative than those used for commercial/industrial receptors. For example, residential dwellings often have less volume per ground surface area and air exchange rates in residential buildings are lower than those in many types of commercial/industrial buildings (Michigan Department of Environmental Quality, 1998). For these reasons and under most circumstances, air concentrations modeled based on the assumptions used for a residential dwelling would be protective of indoor workers as well. Note that none of the WWTP or AP sites currently have any structures that could serve as a consistent indoor occupational workplace.

Estimating indoor airborne concentrations from subsurface soil can be considered to consist of three separate steps:

- Estimating VOC concentration in soil gas at source of contamination (C_{source})
- Estimating an attenuation coefficient that captures the decline in VOC concentration between soil gas at the source and indoor air (á)
- Combining C_{source} and á to estimate VOC concentration in indoor air in the building ($C_{building}$).

An "infinite source" assumption is selected to maintain consistency with the EPA (1996) methodology for particulate emission factor (PEF) and to impart a conservative bias to the evaluation. It is assumed that both the source of VOC contamination in subsurface soil and the foundation of the building are located above the groundwater saturation zone. It is also assumed that VOC contamination in soil does not exist in a nonaqueous phase. Because of the strongly conservative bias imparted by the infinite source assumption, average values are selected for model variables, when possible, if site-specific data are not available. Default values are taken preferentially from EPA (1996) to maintain consistency with the other models described in Section 3.2.2, then from EPA (1997b). The calculations may be performed in the BHHRA using the EPA (2004d) vapor intrusion model which is adapted from of the Johnson and Ettinger (1991).

The first step in estimating indoor air concentrations is to relate the concentration of VOC in soil gas at the source of contamination to the concentration of VOC in soil, as follows (EPA, 2004d):

Eq. 3.4

$$C_{source} = \frac{(H')(C_{so})(P_b)(CF)}{\theta_w + (K_d)(P_b) + (H')(\theta_a)}$$

where:

C_{source}	=	VOC concentration in soil gas at source of contamination (grams per cubic centimeter [g/cm^3], calculated)
H'	=	dimensionless Henry's law constant at average soil temperature (chemical-specific, may be estimated as H · 41 [EPA, 1996])
H	=	Henry's law constant (atmospheres per cubic meter [atmosphere-m^3]/mole, chemical-specific)
C_s	=	contaminant concentration in soil (mg/kg)
ρ_b	=	dry soil bulk density (1.5 g/cm^3, default [EPA, 1996], or site-specific)
CF	=	conversion factor (10^{-6} kg/mg)
θ_w	=	water-filled soil porosity (0.15 L_{water}/L_{soil}, default [EPA, 1996], or site-specific)
K_d	=	soil-water partition coefficient (cubic meters per gram [cm^3/g], chemical-specific, may be estimated as $K_{oc} \cdot f_{oc}$)
K_{oc}	=	soil organic carbon-water partition coefficient (cm^3/g, chemical-specific)
f_{oc}	=	organic carbon content of soil (0.006 grams per gram, default [EPA, 1996], or site-specific)
θ_a	=	air-filled soil porosity (0.28 unitless, default [EPA, 1996], or site-specific estimated as n-θ_w)
n	=	total soil porosity (0.43 unitless, default [EPA, 1996], or site-specific estimated as 1-[ρ_b/ρ_s]).

The next step in calculating indoor air concentrations is the estimation of an attenuation coefficient that reflects the phenomena that reduce the concentration in air between the source and the interior of the building. Because of the many phenomena involved, it is helpful to break this step into several smaller segments.

Diffusion is probably the most important phenomenon involved in the transport of VOC vapors from source to building. The EPA (2004d) modification of the Johnson and Ettinger model provides for multiple layers; i.e., different soil types, each of which would have its own physical properties that affect diffusion between the contaminant source and the foundation of the building. For the purposes of this evaluation, it is simplistically assumed that only one soil type—the predominant soil type in the area—intervenes between source and building foundation.

The equation for effective diffusivity through the soil between the source and the building foundation is given as:

Eq. 3.5

$$D^{eff} = D_a(\theta_a^{3.33} / n^2) + (D_w / H')(\theta_w^{3.33} / n^2)$$

where:

D^{eff}	=	effective diffusion coefficient across capillary zone (cm^2/second, calculated)
D_a	=	diffusivity in air (square centimeters per second [cm^2/second], chemical specific)
θ_a	=	air-filled capillary zone soil porosity (0.28 unitless, default [EPA, 1996], or site-specific, estimated as $n-\theta_w$)
n	=	total capillary zone soil porosity (0.43 unitless, default [EPA, 1996], or site-specific, estimated as $1-[\rho_b/\rho_s]$)
D_w	=	diffusivity in water (cm^2/second, chemical specific)
H'	=	dimensionless Henry's law constant at average soil temperature (chemical-specific, may be estimated as H · 41 [EPA, 1996])
H	=	Henry's law constant (atmosphere-m^3/mole, chemical-specific)
θ_w	=	water-filled capillary zone soil porosity (0.15 L_{water}/L_{soil}, default [EPA, 1996], or site-specific).

The equation for the attenuation coefficient is given as:

Eq. 3.6

$$\alpha = \frac{\left(\left(\frac{D^{eff} A_B}{Q_{building} L_T}\right) x \exp\left(\frac{Q_{soil} L_{crack}}{D^{crack} A_{crack}}\right)\right)}{\left(\exp\left(\frac{Q_{soil} L_{crack}}{D^{crack} A_{crack}}\right) + \left(\frac{D^{eff} A_B}{Q_{building} L_T}\right) + \left(\frac{D^{eff} A_B}{Q_{soil} L_T}\right)\left(\exp\left(\frac{Q_{soil} L_{crack}}{D^{crack} A_{crack}}\right) - 1\right)\right)}$$

where:

α	=	attenuation coefficient (unitless, calculated)
D^{eff}	=	effective diffusion coefficient across soil (cm^2/second)
A_B	=	area of enclosed space below grade (1.51E+6 cm^2, see below)
$Q_{building}$	=	building ventilation rate (4.61E+4 cm^3/second, see below)
L_T	=	distance from source to building (site-specific)
Q_{soil}	=	flow rate of soil gas into enclosed space (cm^2/second, see below)
L_{crack}	=	foundation or slab thickness (15 cm, default [EPA, 1997b])
D^{crack}	=	effective diffusion coefficient through cracks (cm^2/second, assumed to be equivalent to D^{eff} [EPA, 1997b])
A_{crack}	=	area of total cracks (492 cm^2, see below).

EPA (1997a) reviewed several studies of the volumes of houses and recommends 369 m^3 as a central estimate of the volume of a house. Assuming the house has 8-foot (2.44 meters) ceilings and exists on one level, an area of 151.3 square meters, equivalent to 1.51E+6 cm^2, can be estimated as an upper bound on the area below grade.

An average building ventilation rate of 3,984 m^3/day was estimated for a home (EPA, 1997a), which is equivalent to 4.61E+4 cm^3/second.

EPA (2004d) assumes that the only crack available for the entry of soil gas is a 0.1 centimeter (cm)-wide gap at the interface of the floor and foundation. As noted above, it is assumed that the area of the basement floor is 151.3 square meters. Assuming that the house is square, the length of one side would be 12.3 meters, and the total length of the wall would be 49.2 meters (4,920 centimeters). Therefore, the area of the crack would be 492 cm^2. The equation for the flow rate of soil gas into enclosed space is:

Eq. 3.7

$$Q_{soil} = \frac{2\Pi(\Delta P)(k_v)(X_{crack})}{\mu \ln[2(Z_{crack})/(r_{crack})]}$$

where:

Q_{soil}	=	flow rate of soil gas into enclosed space (cm^2/second, calculated)
ΔP	=	pressure differential between soil surface and enclosed space (20 g/cm-second^2)
k_v	=	soil vapor permeability (cm^2, see below)
X_{crack}	=	floor-wall seam perimeter (4,920 cm, see above)
μ	=	viscosity of air (1.83E+5 g/cm-second [EPA, 1992b])
Z_{crack}	=	crack depth below grade (108 cm, see below)
r_{crack}	=	equivalent crack radius (0.1 cm, see below).

Data from which to estimate the crack depth below grade were not located. Presumably, however, houses or other buildings may be built on slabs or on full foundations. EPA (1997b) provides default depths of 15 cm for buildings on slabs and 200 cm for buildings on foundations. The average, 108, is chosen for this evaluation.

Equation 3.7 assumes that vapor transport occurs solely by pressure-driven air flow to an idealized cylinder buried some distance (Z_{crack}) below grade. The length of the cylinder is assumed to be equal to X_{crack}. Therefore, the equivalent crack radius can be estimated as follows:

Eq. 3.8

$$r_{crack} = \eta(\frac{A_B}{X_{crack}})$$

where:

r_{crack}	=	equivalent crack radius (cm, calculated)
η	=	A_{crack}/A_B
A_{crack}	=	area of total cracks (492 cm^2, see above)
A_B	=	area of enclosed space below grade (1.51E+6 cm^2, see above)
X_{crack}	=	floor-wall seam perimeter (4920 cm, see above).

From the foregoing, a value of 0.1 cm is estimated for r_{crack}.

Soil vapor permeability is a very sensitive parameter associated with convective transport of vapors within the zone of influence of a building (EPA, 2004d). It can be estimated as the product of soil intrinsic permeability and the relative air permeability at the estimated water-filled soil porosity (θ_w). Soil intrinsic permeability is estimated as follows:

Eq. 3.9

$$k_i = \frac{K_s \mu_w}{\rho_w g}$$

where:

k_i	=	soil intrinsic permeability (cm^2, calculated)
K_s	=	soil saturation hydraulic conductivity (cm/second, see below)
μ_w	=	dynamic viscosity of water (0.01307 g/cm-second [EPA, 1997b])
ρ_w	=	density of water (0.999 g/cm^2, [EPA, 1997b])
g	=	acceleration due to gravity (980.665 cm/$second^2$ [EPA, 1997b]).

Soil saturation hydraulic conductivity is related to soil texture. Site-specific data will be used in conjunction with Table 4 of EPA (1997b) to estimate an approximate value for K_s.

Relative air permeability is estimated as follows:

Eq. 3.10

$$k_{rg} = \left(1 - S_{te}\right)^{0.5}\left(1 - S_{te}^{1/M}\right)^{2M}$$

where:

k_{rg}	=	relative air permeability (positive unitless value, calculated)
S_{te}	=	effective total fluid saturation (unitless, see below)
M	=	van Genuchten shape parameter (unitless, see below).

Site-specific data regarding the nature of the soil will be used in conjunction with Table 2 of EPA (1997b) to estimate an appropriate van Genuchten shape parameter.

S_{te} is calculated as follows:

Eq. 3.11

$$S_{te} = \frac{\theta_w - \theta_r}{n - \theta_r}$$

where:

S_{te}	=	effective total fluid saturation (unitless, calculated)
θ_w	=	water-filled soil porosity (0.15 L_{water}/L_{soil}, default [EPA, 1996], or site-specific)
θ_r	=	soil water content (cm^3/cm^3, taken from Table 2 of EPA [1997b])
n	=	total soil porosity (0.43 unitless, default [EPA, 1996], or site-specific estimated as 1-[ρ_b/ρ_s]).

Soil vapor permeability is estimated as follows:

Eq. 3.12

$$k_v = (k_i)(k_{rg})$$

where:

k_v	=	soil vapor permeability (cm^2, calculated)
k_i	=	soil intrinsic permeability (cm^2)
k_{rg}	=	relative air permeability (unitless).

The foregoing equation permits calculation of the attenuation coefficient, which, in turn, permits calculation of the concentration of VOC in indoor air in the building, as follows:

Eq. 3.13

$$C_{building} = \alpha(CF)(C_{source})$$

where:

$C_{building}$ = VOC concentration in indoor air in the building (mg/m^3, calculated)
α = attenuation coefficient (unitless)
CF = conversion factor (1E+9 mg-cm^3/g-m^3)
C_{source} = VOC concentration in soil gas at source of contamination (g/cm^3).

3.2.2.3 VOC Concentrations from Subsurface Soil in Ambient Air

The construction worker may be exposed to VOCs released from subsurface soil by volatilization. EPCs of VOCs in ambient air due to volatilization are estimated with a chemical-specific soil volatilization factor calculated from the following equations and defaults provided by EPA (1996):

Eq. 3.14

$$VF_s = Q/C \times CF \times \left(\frac{[\ 3.14 \times D_A \times T\ ]^{1/2}}{2 \times \rho_b \times D_A} \right)$$

Eq. 3.15

$$D_A = \frac{(\theta_a^{10/3} \times D_i \times H' + \theta_w^{10/3} \times D_W\ ) / n^2}{\rho_b \times K_d + \theta_w + \theta_a \times H'}$$

where:

VF_s = chemical-from-soil volatilization factor (m^3/kg, calculated)
Q/C = inverse of the mean concentration at center of square source (55.99 g/m^2-second per kg/m^3, site-specific value from Table 3 of EPA [1996] [Zone 5, Cleveland, 5-acre site])
CF = conversion factor (1E-4 m^2/cm^2)
D_A = apparent diffusivity (cm^2/second, calculated)
T = exposure interval (seconds, receptor-specific, estimated as ED · 3.15E7 seconds/year)
ED = exposure duration (years, receptor-specific)
ρ_b = dry soil bulk density (1.5 g/cm^3, default, or site-specific)
θ_a = air-filled soil porosity (0.28 unitless, default, or site-specific estimated as n-θ_w)
n = total soil porosity (0.43 unitless, default, or site-specific estimated as 1-[ρ_b/ρ_s])
ρ_s = true soil or particle density (2.65 g/cm^3, default, or site-specific)
θ_w = water-filled soil porosity (0.15 L_{water}/L_{soil}, default, or site-specific)
D_i = diffusivity in air (cm^2/second, chemical specific)
H' = dimensionless Henry's law constant (chemical-specific, may be estimated as H · 41)
H = Henry's law constant (atmosphere-m^3/mole, chemical-specific)

D_W = diffusivity in water (cm^2/second, chemical-specific)
K_d = soil-water partition coefficient (cm^3/g, chemical-specific, may be estimated as $K_{oc} \cdot f_{oc}$)
K_{oc} = soil organic carbon-water partition coefficient (cm^3/g, chemical-specific)
f_{oc} = organic carbon content of soil (6E-3 g/g, default, or site-specific).

The concentration of COPC in ambient air is estimated as follows:

Eq. 3.16

$$C_a = \frac{C_{so}}{VF}$$

where:

C_a = contaminant concentration in air (mg/m^3, calculated)
C_s = contaminant concentration in soil (mg/kg)
VF = chemical-from-soil volatilization factor (m^3/kg, chemical-specific, calculated in Eq. 3.14).

3.2.2.4 Concentrations in Household Air from Groundwater Use

The inhalation of VOCs released from groundwater, which is assumed to be used as tap water, is evaluated for the on-site residential scenario. Chemicals that have a Henry's Law value exceeding 1E-5 atmospheres per cubic meter atm/m^3/mole and a molecular weight less than 200 grams per mole are considered to be VOCs and are subject to evaluation via this pathway; Henry's Law values and molecular weights will be presented in table format with appropriate references. Other groundwater contaminants are considered on a case by case basis for their potential contribution to risk via the inhalation pathway based on the degree of departure from the Henry's Law and molecular weight criteria, groundwater concentration, and toxicity.

The simple whole-house, tap water-to-air model described in Part B of the Human Health Evaluation Manual (HHEM) (EPA, 1991b) was used to evaluate the tap water-to-air pathway. This model was selected based on correspondence between OEPA (2004) and USACE. Part B of the HHEM recommends a volatilization constant of 0.0005 for the total concentrations of all VOCs detected in groundwater; the conversion is characterized by the following equation:

Eq. 3.17

$$C_a = C_{gw} \times K_{wa} \times 1{,}000 \, {}^{L}\!/_{m^3}$$

where:

C_a = modeled concentration in air (mg/m^3)
C_{gw} = groundwater EPC (mg/L)
K_{wa} = tap water-to-air volatilization constant (0.0005 [unitless]: [EPA, 1991b])

Implicit in the HHEM Part B application of this model are the following: 1) a family of four uses the groundwater as the sole source of household tap water; 2) the volume of the house is 150 m^3; 3) the daily groundwater use is 720 L/day; 4) 50 percent of VOCs in tap water volatilize to household air; and 5) the air exchange rate of the house is 0.25 volumes per hour (EPA, 1991b). The EPA (1997a) *Exposure Factors Handbook* lists values different from some of those assumed by HHEM Part B.

3.2.2.5 Concentrations of VOCs in Groundwater: Resident Dermal Uptake

Volatilization of VOCs from household water reduces the remaining concentration available for dermal contact. As mentioned in Section 3.2.2.4, the HHEM Part B whole-house tap water-to-air model assumes that 50 percent of the VOC concentrations are released to household air. Thus, the concentrations of VOCs remaining in the water after volatilization occurs are calculated by difference as follows:

Eq. 3.18

$$C_d = C_{gw} \times (1 - F_v)$$

where:

C_d = concentration of VOC in household water available for dermal exposure (mg/L, calculated)
C_{gw} = concentration of VOC in groundwater (mg/L)
F_v = fraction of VOCs volatilized to air, (0.5 unitless).

Only the concentration remaining in tap water after volatilization (C_d), as applicable, is assumed to be available for contact with the skin during bathing/showering.

3.2.2.6 Exposure-Point Concentrations of COPCs in Venison

The hunter is assumed to harvest and consume game and share it with family members, including small children. The game is assumed to be venison, because the white-tailed deer is the species hunted most widely and most likely to provide a regular contribution to the diet. Data do not exist to reliably estimate contaminant concentrations in venison, but the following simplifying assumptions permit estimates sufficient for a BHHRA.

- Deer are small ruminants and, as such, are not unlike cattle; thus, it is reasonable to assume they may have similar physiological processes that could yield similar biotransfer factors. Unlike beef, however, deer meat does not undergo marbling with fat, and deer fat is quite unpalatable and is likely to be trimmed rather than consumed. Therefore, the biotransfer factors for edible venison are derived by adjusting biotransfer factors for beef to account for differences in the fat content of table-ready beef (cooked choice retail cuts trimmed to 0 inches of fat: average 14.4 percent fat) and venison (cooked boneless muscle meats: average 2.9 percent fat) (Nutrient Database, 1997).

- Deer are expected to browse a much larger area than that encompassed in any of the WWTP or AP sites; therefore, the fraction of total browse consumed from the contaminated site is expected to be small.

- Indirect food chain pathways may be significant for some metals and for those SVOCs that persist in the environment and have the tendency to bioaccumulate. VOCs are generally mobile in the environment and labile in biological systems and do not tend to bioaccumulate.

To reflect the assumptions previously noted, venison biotransfer factors are estimated by multiplying beef biotransfer factors by 2.9/14.4 (or 0.20), and by a fraction, FI_r. FI_r reflects the areal portion of the site compared to a deer's home range area. These assumptions are captured in the following equation:

Eq. 3.19

$$B_v = 0.20(B_b)(FI_r)$$

where:

B_v	=	biotransfer factor for venison (unitless, calculated)
0.20	=	factor to reflect differences in fat content between beef and venison (0.20, unitless, see above)
FI_r	=	areal portion of site compared to a deer's home range (0.03, unitless, see below)
B_b	=	biotransfer factor for beef.

Values for B_b for metals will be provided in the toxicity profiles appended to the BHHRA. Toxicity profiles will be prepared for each of the COPCs evaluated in the BHHRA. The toxicity profiles briefly describe the uses of the chemical, its physical properties, behavior in environmental media, biotransfer capability, and toxicity values.

The WWTP and AP areas are relatively small in comparison to the home range of a white-tailed deer. Each of these study areas is between 1.5 and 4 acres, whereas the home range of the white-tailed deer is between 150 and 1,280 acres (Sample and Suter, 1994). Even if the low end of this range (150 acres) is assumed for deer in northern Ohio, the largest of the study areas comprises approximately 3 percent of this land area. Therefore, an FI_r value of 0.03 will be used in the BHHRA.

Deer are assumed to be exposed to contaminants by ingesting browse growing on contaminated soil. It is estimated that deer consume approximately 1.74 kg of browse per day (Sample, et al., 1996), which is approximately 50 percent dry matter (DM), or 0.87 kg browse DM per day (Mautz, et al., 1976). The contaminant concentration in browse is estimated from the following equation, which was originally developed for estimating the contaminant concentration in forage to which cattle may be exposed (EPA, 1994):

Eq. 3.20

$$C_p = (CF)(C_s)(B_p)$$

where:

C_p	=	concentration of contaminant in (plant) forage DM (mg/kg, calculated)
CF	=	conversion factor to adjust for soil containing 20 percent moisture (1.25, unitless).
C_s	=	concentration of contaminant in soil (mg/kg)
B_p	=	soil-to-forage biotransfer factor (mg of chemical per kg of dry plant/mg of chemical per kg of dry soil).

Values for B_p will be taken from the toxicity profiles appended to the BHHRA. B_p values for the vegetative parts of plants, rather than the reproductive parts of plants, will be selected, when possible, because deer browse year-round, and the vegetative parts are more available for the greater part of the year.

The concentration of COPC in venison can be estimated from the following equation (adapted from EPA [1994]):

Eq. 3.21

$$C_v = (Q_p)(C_p)(B_v)$$

where:

C_v = contaminant concentration in venison (mg/kg, calculated)
Q_p = browse ingestion rate (0.87 kg DM/day)
C_p = contaminant concentration in browse DM (mg/kg)
B_v = biotransfer factor for venison (days/kg).

3.3 Quantification of Chemical Intake

This section describes the models used to quantify doses or intakes of the COPCs by the exposure pathways identified above. Models were taken or modified from EPA (1989a) unless otherwise indicated.

3.3.1 Inhalation of COPCs in Air

The following equation is used to estimate the inhaled dose of COPC in air (groundskeeper, construction worker, on-site resident: inhalation of dust and VOCs in ambient air from surface or total soil; construction worker: inhalation of VOCs in ambient air from subsurface soil; indoor worker and on-site resident: inhalation of VOCs in indoor air from subsurface soil):

Eq. 3.22

$$I_a = \frac{(C_a)(FI_a)(IR_a)(EF)(ED)}{(BW)(AT)}$$

where:

I_a = inhaled dose of COPC (milligrams per kilograms per day [mg/kg-day], calculated)
C_a = concentration of COPC in air (mg/m^3)
FI_a = fraction of exposure attributed to site media (unitless)
IR_a = inhalation rate (m^3/day)
EF = exposure frequency (days/year)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (days).

3.3.2 Incidental Ingestion of COPCs in Soil

The ingested dose of COPC in soil (groundskeeper, construction worker, on-site resident, indoor worker, hunter) is estimated from the equation:

Eq. 3.23

$$I_s = \frac{(C_s)(FI_s)(IR_s)(EF)(ED)(CF)}{(BW)(AT)}$$

where:

I_s	=	ingested dose of COPC in soil (mg/kg-day, calculated)
C_s	=	concentration of COPC in soil (mg/kg)
FI_s	=	fraction of exposure attributed to site soil (unitless)
IR_s	=	ingestion rate of soil (mg/day)
EF	=	exposure frequency (days/year)
ED	=	exposure duration (years)
CF	=	conversion factor (1E-6 kg/mg)
BW	=	body weight (kg)
AT	=	averaging time (days).

3.3.3 Incidental Ingestion of COPCs in Sediment

The ingested dose of COPC in sediment (construction worker, on-site resident) is estimated from the equation:

Eq. 3.24

$$I_{sd} = \frac{(C_{sd})(FI_{sd})(IR_{sd})(EF)(ED)(CF)}{(BW)(AT)}$$

where:

I_{sd}	=	ingested dose of COPC in sediment (mg/kg-day, calculated)
C_{sd}	=	concentration of COPC in sediment (mg/kg)
FI_{sd}	=	fraction of exposure attributed to site sediment (unitless)
IR_{sd}	=	ingestion rate of sediment (mg/day)
EF	=	exposure frequency (days/year)
ED	=	exposure duration (years)
CF	=	conversion factor (1E-6 kg/mg)
BW	=	body weight (kg)
AT	=	averaging time (days).

3.3.4 Dermal Contact with COPCs in Soil, Sediment, or Water

Unlike the methodologies for estimating inhaled or ingested doses of COPC, which quantify the dose presented to the barrier membrane (the pulmonary or gastrointestinal mucosa, respectively), dermal dose is estimated as the dose that crosses the skin and is systemically absorbed. For this reason, dermal toxicity values are also based on absorbed dose. The absorbed dose of COPC is estimated from the following equation (EPA, 2004a):

Eq. 3.25

$$DAD = \frac{(DA)(SA)(EF)(ED)}{(BW)(AT)}$$

where:

DAD = average dermally absorbed dose of COPC (mg/kg-day, calculated)
DA = dose absorbed per unit body surface area per day (mg/cm^2-day)
SA = SA_s for soil, SA_{sd} for sediment, SA_{sw} for surface water, = surface area of the skin exposed (cm^2)
EF = exposure frequency (days/year)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (days).

DA is calculated differently for dermal uptake from soil or sediment and from water. Dermal uptake of constituents from soil (groundskeeper, construction worker, on-site resident, hunter) or sediment (construction worker, on-site resident) assumes that absorption is a function of the fraction of a dermally applied dose that is absorbed. It is calculated from the following equation (EPA, 2004a):

Eq. 3.26

$$DA = (C)(FI)(CF)(AF)(ABS)$$

where:

DA = dose absorbed per unit body surface area per day (mg/cm^2-day, calculated)
C = C_s for soil, C_{sd} for sediment, = concentration of COPC in medium (mg/kg)
FI = FI_s for soil, FI_{sd} for sediment, = fraction of exposure attributed to site medium (unitless)
CF = conversion factor (1E-6 kg/mg)
AF = AF_s for soil, AF_{sd} for sediment, = soil- or sediment-to-skin adherence factor (mg/cm^2-day)
ABS = absorption fraction (unitless, chemical-specific).

ABS values will be provided in the toxicity profiles for each COPC that will be appended to the BHHRA.

Quantification of dermal uptake of constituents from surface water (construction worker, on-site resident) depends on a K_p, which describes the rate of movement of a constituent from water

across the dermal barrier to the systemic circulation (EPA, 2004a). The equation for dermal uptake of chemicals from water is the same as the equation for dermal uptake of chemicals from soil (Eq. 3.25). DA is calculated differently for inorganic and organic chemicals in water. For inorganic chemicals, DA is calculated from the following equation:

Eq. 3.27

$$DA = (C)(FI)(K_p)(ET)(CF)$$

where:

DA	=	dose absorbed per unit body surface area per day (mg/cm^2-day, calculated)
C	=	concentration of COPC in water (mg/L); C_{sw} for surface water; C_{gw} = for groundwater
K_p	=	permeability coefficient (cm/hour)
ET	=	ET_{sw} for surface water = time of exposure (hours/day)
CF	=	conversion factor (1E-3 L/cm^3).

K_p for organic chemicals varies by several orders of magnitude and is highly dependent on lipophilicity, expressed as a function of the octanol/water partition coefficient (EPA, 2004a). Because the stratum corneum (the outer skin layer) is rich in lipid content, it may act as a sink, initially reducing the transport of chemical to the systemic circulation. With continued exposure and the attainment of steady-state conditions, the rate of dermal uptake increases. Therefore, different equations are used to estimate DA, depending on whether the exposure time is less than or greater than the estimated time to reach steady state. Non-steady-state exposures occur when either the exposure time is relatively brief (e.g., showering, for most chemicals) or when intermittent exposure occurs throughout the day (e.g., wading exposure to surface water or washing of hands). For exposure scenarios under which steady state is not reached for a given organic chemical (τ>exposure time [ET], see below), the following equation is used to calculate DA (EPA, 2004a):

Eq. 3.28

$$DA = 2(FA)(K_p)(C_w)(CF)\sqrt{\left(\frac{6\tau(ET_w)}{\pi}\right)}$$

where:

DA	=	dose absorbed per unit body surface area per day (mg/cm^2-day, calculated)
C_w	=	concentration of COPC in water (mg/L); C_{sw} for surface water; C_{gw} = for groundwater
FA	=	fraction absorbed from the water (unitless)
K_p	=	permeability coefficient (cm/hour)
CF	=	conversion factor (1E-3 L/cm^3)
τ	=	time for concentration of contaminant in stratum corneum to reach steady state per event (hours)
ET	=	time of exposure (hour(s)/day); ET_{sw} for surface water; ET_{gw} = for groundwater.

In cases where steady state is reached (τ<ET), such as where the duration of a bath exceeds the time to reach steady state for a given organic compound, the following equation is used to calculate DA (EPA, 2004a):

Eq. 3.29

$$DA = (FA)(K_p)(C_w)(CF)\left[\frac{ET}{1+B} + 2\tau\left(\frac{1+3B+3B^2}{(1+B)^2}\right)\right]$$

where:

DA	=	dose absorbed per unit body surface area per day (mg/cm^2-day, calculated)
C_w	=	concentration of COPC in water (mg/L); C_{sw} for surface water; C_{gw} = for groundwater
FA	=	fraction absorbed from the water (unitless)
K_p	=	permeability coefficient (cm/hour)
CF	=	conversion factor (1E-3 L/cm^3)
τ	=	time for concentration of contaminant in stratum corneum to reach steady state per event (hours)
ET	=	time of exposure (hour(s)/day); ET_{sw} for surface water; ET_{gw} = for groundwater
B	=	Ratio of the permeability coefficient of a compound through the stratum corneum relative to its permeability coefficient across the viable epidermis (unitless).

Assuming one exposure event/day allows expressing ET as hour(s)/day, which preserves the dimensional integrity of the equation.

When available, values for K_p and τ are taken from EPA (2004a). For organics that have no K_p values listed, the values are calculated using the following equation (EPA, 2004a):

Eq. 3.30

$$\text{Log}(K_p) = -2.80 + 0.66(\log K_{ow}) - 0.0056(MW)$$

where:

K_p = permeability coefficient (cm/hour, calculated)
log K_{ow} = log of the octanol/water partition coefficient (unitless)
MW = molecular weight.

Where values for τ are not available, they were calculated using the following equation (EPA, 2004a).

Eq. 3.31

$$\tau = 0.105 \times 10^{(0.0056 \times MW)}$$

where:

τ = time for concentration of contaminant in stratum corneum to reach steady state (hours, calculated)
MW = molecular weight.

Values of K_p and τ to be used in the BHHRA will be summarized in a table of the BHHRA. The values will be documented in toxicity profiles appended to the BHHRA.

3.3.5 Consumption of Venison

Consumption of venison by the hunter or the hunter's child is evaluated by the following equation:

Eq. 3.32

$$I_v = \frac{(C_v)(IR_v)(EF)(ED)}{(BW)(AT)}$$

where:

IR_v = ingested dose of COPC in venison (mg/kg-day, calculated)
Cv = concentration of COPC in venison (mg/kg)
IRv = venison ingestion rate (kg/day)
EF = exposure frequency (days/year)

ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (days).

4.0 Toxicity Evaluation

Toxicity is defined as the ability of a chemical to induce adverse effects in biological systems. The purpose of the toxicity assessment is two-fold:

- Identify the cancer and noncancer effects that may arise from exposure of humans to the COPC (hazard assessment)
- Provide an estimate of the quantitative relationship between the magnitude and duration of exposure and the probability or severity of adverse effects (dose-response assessment).

The latter is accomplished by the derivation of cancer and noncancer toxicity values, as described in the following sections.

4.1 Evaluation of Carcinogenicity

A few chemicals are known, and many more are suspected, to be human carcinogens. The evaluation of the potential carcinogenicity of a chemical includes both a qualitative and a quantitative aspect (EPA, 2005). The qualitative aspect is a weight-of-evidence evaluation of the likelihood that a chemical might induce cancer in humans. EPA (2005) recognizes five weight-of-evidence group classifications for carcinogenicity. Formerly, EPA (1986) used a letter-based system to describe the weight of evidence for carcinogenicity. Reference to this former system is included because many of the carcinogenicity assessments listed on the Integrated Risk Information System (IRIS) use the former letter-based system. The five EPA weight-of-evidence classifications are as follows:

- ***Carcinogenic to Humans*** (corresponds to the former Group A – Human Carcinogen).
- ***Likely to be Carcinogenic to Humans*** (Includes both the former Group B1 and Group B2 Probable Human Carcinogens)
- ***Suggestive Evidence of Carcinogenic Potential*** (corresponds to the former Group C – Possible Human Carcinogen)
- ***Inadequate Information to Assess Carcinogenic Potential*** (corresponds to the former Group D - Not Classifiable as to Human Carcinogenicity)
- ***Not Likely to be Carcinogenic to Humans*** (corresponds to the former Group E - Evidence of Noncarcinogenicity to Humans)

The toxicity value for carcinogenicity, called a cancer slope factor (SF), is an estimate of potency. SFs are developed only for chemicals in the first three groups and only if the data are sufficient. The SFs are statistically derived from the dose-response curve from the best human or animal study or studies of the chemical. Although human data are often considered to be more reliable than animal data because there is no need to extrapolate the results obtained in one species to another, most human studies have one or more of the following limitations:

- The duration of exposure is usually considerably less than lifetime.
- The concentration or dose of chemical to which the humans were exposed can be approximated only crudely, usually from historical data.
- Concurrent exposure to other chemicals frequently confounds interpretation.
- Data regarding other factors (tobacco, alcohol, illicit or medicinal drug use, nutritional factors and dietary habits, heredity) are usually insufficient to eliminate confounding or quantify confounding effects on the results.
- Most epidemiologic studies are occupational investigations of workers, which may not accurately reflect the range of sensitivities of the general population.
- Most epidemiologic studies lack the statistical power (i.e., sample size) to detect a low, but chemical-related increased incidence of tumors.

Most potency estimates are derived from animal data, which present different limitations:

- It is necessary to extrapolate from results in animals to predict results in humans, usually by estimating an equivalent human dose from the animal dose.
- The range of sensitivities arising from genotypic and phenotypic diversity in the human population is not reflected in the animal models ordinarily used in cancer studies.
- Usually very high doses of chemical are used, which may alter normal biology, creating a physiologically artificial state and introducing substantial uncertainty regarding the extrapolation to the low-dose range expected with environmental exposure.
- Individual studies vary in quality (e.g., duration of exposure, group size, scope of evaluation, adequacy of control groups, appropriateness of dose range, absence of concurrent disease, sufficient long-term survival to detect tumors with long induction or latency periods).

The SF is usually expressed as "extra risk" per unit dose, that is, the additional risk above background in a population corrected for background incidence. It is calculated using the following equation:

Eq. 4.1

$$SF = (p_{(d)} - p_{(0)})/(1 - p_{(0)})$$

where:

SF = chemical-specific slope factor per unit dose $(mg/kg\text{-}day)^{-1}$

$p_{(d)}$ = the probability of developing cancer at a dose of 1 mg/kg-day of the chemical of interest

$p_{(0)}$ = the background probability of developing cancer at a dose of 0 mg/kg-day of the chemical of interest.

The SF is expressed as risk per mg/kg-day, shown mathematically as $(mg/kg\text{-}day)^{-1}$. To be appropriately conservative, the SF is usually the 95 percent upper bound on the slope of the dose-response curve extrapolated from high (experimental) doses to the low-dose range expected in environmental exposure scenarios. EPA (1986) assumes that there are no thresholds for carcinogenic expression; therefore, any exposure represents some quantifiable risk, however miniscule it may be.

The oral SF is usually derived directly from the experimental dose data, because oral dose is usually expressed as mg/kg-day. When the test chemical was administered in the diet or drinking water, oral dose first must be estimated from data for the concentration of the test chemical in the food or water, food or water intake data, and body weight data.

The Integrated Risk Information System (IRIS) (EPA, 2009d) expresses inhalation cancer potency as a unit risk based on concentration, or risk per microgram of chemical per m^3 of ambient air, shown mathematically as $(\mu g/m^3)^{-1}$. Because cancer risk characterization requires an SF expressed as risk per mg/kg-day, the unit risk must be converted to the mathematical equivalent of an inhalation cancer SF, or risk per unit dose as $(mg/kg\text{-}day)^{-1}$. Because the inhalation unit risk is based on continuous lifetime exposure of an adult human (assumed to inhale 20 m^3 of air per day and to weigh 70 kg) the mathematical conversion consists of multiplying the unit risk (per $\mu g/m^3$) by 70 kg and by 1,000 µg/mg, and dividing the result by 20 m^3 per day.

4.2 Evaluation of Noncarcinogenic Effects

Many chemicals, whether or not associated with carcinogenicity, are associated with adverse noncarcinogenic effects. The evaluation of noncancer effects (EPA, 1989b) involves the following:

- Qualitative identification of the adverse effect(s) associated with the chemical; these may differ depending on the duration (acute or chronic) or route (oral or inhalation) of exposure.
- Identification of the critical effect for each duration of exposure (i.e., the first adverse effect that occurs as dose is increased).
- Estimation of the threshold dose for the critical effect for each duration of exposure.
- Development of an uncertainty factor (UF); i.e., quantification of the uncertainty associated with interspecies extrapolation, intraspecies variation in sensitivity, severity of the critical effect, slope of the dose-response curve, and deficiencies in the database, in regard to developing a reference dose (RfD) for human exposure.
- Identification of the target organ(s) for the critical effect for each route of exposure.

These information points are used to derive an exposure route- and duration-specific toxicity value called an RfD, expressed as mg/kg-day, which is considered to be the dose for humans, with uncertainty of an order of magnitude or greater, at which adverse effects are not expected to occur. Mathematically, it is estimated as the ratio of the threshold dose to the UF. For purposes of risk assessment, chronic exposure is typically defined as equal to or greater than 7 years, i.e., at least 10 percent of expected life span; subchronic exposure is typically defined as 2 weeks to 7 years. However, professional judgment may be used where exposure durations approach 10 percent of the expected life span. Also, exposure during a critical stage of development, such as a portion of early childhood, may be treated as chronic even if the anticipated exposure duration were to be considerably less than 10 percent of the expected life span.

IRIS (EPA, 2009d) expresses the inhalation noncancer reference value as a reference concentration (RfC) in units of mg/m^3. Because noncancer risk characterization requires a reference value expressed as mg/kg-day, the RfC must be converted to an inhalation RfD. Because the inhalation RfC is based on continuous exposure of an adult human (assumed to inhale 20 m^3 of air per day and to weigh 70 kg), the mathematical conversion consists of multiplying the RfC (mg/m^3) by 20 m^3/day and dividing the result by 70 kg.

RfD and RfC values are derived for both chronic and subchronic exposure. Under the assumption of monotonicity (incidence, intensity, or severity of effects can increase, but cannot decrease, with increasing magnitude or duration of exposure), a chronic RfD may be considered sufficiently protective for subchronic exposure, but a subchronic RfD may not be protective for chronic exposure. Currently, subchronic RfD values exist for few chemicals. Subchronic RfD values can be derived from chronic RfD values as follows:

- If the UF applied in the derivation of the chronic RfD (or RfC) does not provide for expansion from subchronic to chronic exposure (e.g., if the chronic RfD was derived from a chronic study), the chronic RfD is adopted as being sufficiently protective for subchronic exposure.

- If the UF applied in the derivation of the chronic RfD (or RfC) contains a component to expand from subchronic to chronic exposure, the subchronic RfD is derived by multiplying the chronic RfD by the factor used to expand from subchronic to chronic exposure (e.g., if a factor of 10 was used to expand from subchronic to chronic exposure, the subchronic RfD would be 10 times larger than the chronic RfD).

Only chronic RfDs and RfCs will be used in the risk characterization of the BHHRA.

4.3 Dermal Toxicity Values

Dermal RfDs and SFs are derived from the corresponding oral values, provided there is no evidence to suggest that dermal exposure induces exposure route-specific effects that are not appropriately modeled by oral exposure data. In the derivation of a dermal RfD, the oral RfD is multiplied by the gastrointestinal absorption factor (GAF), expressed as a decimal fraction. The resulting dermal RfD, therefore, is based on absorbed dose. The RfD based on absorbed dose is the appropriate value with which to compare a dermal dose, because dermal doses are expressed as absorbed doses rather than exposure doses. The dermal SF is derived by dividing the oral SF by the GAF. The oral SF is divided, rather than multiplied, by the GAF because the SF is expressed as a reciprocal dose.

4.4 Target Organ Toxicity

As a matter of science policy, EPA assumes dose and effect to be additive for noncarcinogenic effects (EPA, 1989a). This assumption provides the justification for adding the HQs or HIs in the risk characterization for noncancer effects (Section 5.2) resulting from exposure to multiple chemicals, pathways, or media. However, EPA (1989a) acknowledges that adding all HQ or HI

values may overestimate hazard, because the assumption of additivity is probably appropriate only for those chemicals that exert their toxicity by the same mechanism.

Mechanisms of toxicity data sufficient for predicting additivity with a high level of confidence are available for very few chemicals. In the absence of such data, EPA (1989a) assumes that chemicals that act on the same target organ may do so by the same mechanism of toxicity; that is, the target organ serves as a surrogate for mechanism of toxicity. When total HI for all media for a receptor exceeds 1 due to the contributions of several chemicals, it is appropriate to segregate the chemicals by route of exposure and mechanism of toxicity (i.e., target organ) and estimate separate HI values for each target organ.

As a practical matter, because human environmental exposures are likely to involve near- or sub-threshold doses, the target organ chosen for a given chemical is the one associated with the critical effect. If more than one organ is affected by a given chemical at the threshold, then all affected target organs are selected for this chemical. The target organ is also selected on the basis of duration of exposure (i.e., the target organ for chronic or subchronic exposure to low or moderate doses is selected rather than the target organ for acute exposure to high doses) and route of exposure. Because dermal RfD values are derived from oral RfD values, the oral target organ is adopted as the dermal target organ. For some chemicals, no target organ is identified. This occurs when no adverse effects are observed or when adverse effects such as reduced longevity or growth rate are not accompanied by recognized organ- or system-specific functional or morphologic alteration.

4.5 Sources of Toxicity Information Used in the Risk Assessment

Toxicity values were selected for use in the BHHRA based on EPA Office of Solid Waste and Emergency Response Directive 9285.7-53 (EPA, 2003), which prescribes the following hierarchy:

- **Tier 1** values: IRIS (EPA, 2009d) database.
- **Tier 2** values: These are EPA's provisional peer-reviewed toxicity values. The provisional peer-reviewed toxicity values are developed by the Office of Research and Development, the National Center for Environmental Assessment, and the Superfund Health Risk Technical Support Center on a chemical-specific basis when requested by the Superfund program.
- **Tier 3** values: These are other toxicity values from additional EPA and non-EPA sources of toxicity information. As stated in the EPA Office of Solid Waste and

Emergency Response directive, "priority should be given to those sources of information that are the most current, the basis for which is transparent and publicly available, and which have been peer reviewed." Two common examples of Tier 3 values are the EPA's Health Effects Assessment Summary Tables (EPA, 1997c) and the California Environmental Protection Agency (2005) Office of Environmental Health Hazard Assessment Toxicity Criteria Database.

The Environmental Council of States-Department of Defense (2007) has recently issued a toxicity value hierarchy that basically supports the EPA (2003) hierarchy presented above but places higher emphasis on the necessity for external peer review.

GAFs used to derive dermal RfD values and SFs from the corresponding oral toxicity values are obtained from the following sources:

- Oral absorption efficiency data compiled by the National Center for Environmental Assessment for the Superfund Health Risk Technical Support Center of EPA
- Federal agency reviews of the empirical data, such as Agency for Toxic Substances and Disease Registry toxicological profiles and various EPA criteria documents
- Other published reviews of the empirical data
- The primary literature.

GAFs obtained from reviews are compared to empirical (especially more recent) data, when possible, and are evaluated for suitability for use in deriving dermal toxicity values from oral toxicity values. The suitability of the GAF increases when the following similarities are present in the oral pharmacokinetic study from which the GAF is derived and in the key toxicity study from which the oral toxicity value is derived:

- The same strain, sex, age, and species of test animal were used.
- The same chemical form (e.g., the same salt or complex of an inorganic element or organic compound) was used.
- The same mode of administration (e.g., diet, drinking water, or gavage vehicle) was used.
- Similar dose rates were used.

Individual toxicity profiles will be appended to the BHHRA for all of the COPCs evaluated in the BHHRA. Summary toxicity information sufficient to support the risk calculations, including toxicity values, GAFs, target organs, and sources, will be provided in tables to the BHHRA.

5.0 Risk Characterization

Risk characterization is the process of applying numerical methods and professional judgment to determine the potential for adverse human health effects to result from the presence of site-specific contaminants. This is done by combining the intake rates estimated during the exposure assessment with the appropriate toxicity information identified during the toxicity assessment. Noncancer hazards and cancer risks are characterized separately, including COPCs that induce both types of effects.

Quantitative expressions are calculated during risk characterization that describe the probability of developing cancer (i.e., ILCRs), or the nonprobabilistic comparison of estimated dose with an RfD for noncancer effects (i.e., HQs and HIs). Quantitative estimates are developed for individual chemicals, exposure pathways, and exposure media for each receptor. These quantitative risk characterization expressions, in combination with qualitative information, are used to guide risk management decisions. Risk characterization, as described in this section, is applied only to COPCs.

Generally, the risk characterization follows the methodology prescribed by EPA (1989a), as modified by more recent information and guidance. EPA methods are appropriately designed to be health protective and tend to overestimate rather than underestimate risk. The risk results, however, may be overly conservative, because risk characterization involves multiplication of the conservative assumptions built into the estimation of the EPCs, exposure (intake) estimates, and toxicity dose-response assessments.

5.1 Cancer Risk

The risk from exposure to potential chemical carcinogens is estimated as the probability of an individual developing cancer over a lifetime, and is called the ILCR. In the low-dose range, which would be expected for most environmental exposures, cancer risk is estimated from the following linear equation (EPA, 1989a):

Eq. 5.1

$$ILCR = (CDI)(SF)$$

where:

ILCR = incremental lifetime cancer risk, a unitless expression of the probability of developing cancer, adjusted for background incidence, calculated
CDI = chronic daily intake, averaged over 70 years (mg/kg-day)
SF = cancer slope factor (risk per mg/kg-day).

The chronic daily intake (CDI) term in Equation 5.1 is equivalent to the "I" or "DAD" terms (intake or dose) in Equations 3.22 through 3.25 and 3.32 when these equations are evaluated for cancer intakes.

The use of Equation 5.1 assumes that chemical carcinogenesis does not exhibit a threshold, and that the dose-response relationship is linear in the low-dose range. Because this equation could generate theoretical cancer risks greater than 1 for high-dose levels, it is considered to be inaccurate at cancer risks greater than 1E-2. In these cases, cancer risk is estimated by the one-hit model (EPA, 1989a):

Eq. 5.2

$$ILCR = 1 - e^{[-(CDI)(SF)]}$$

where:

ILCR = incremental lifetime cancer risk, a unitless expression of the probability of developing cancer, adjusted for background incidence, calculated

$-e^{(CDI)(SF)}$ = the exponential of the negative of the risk calculated using Equation 5.1

As a matter of policy, the EPA (1986) considers the carcinogenic potency of simultaneous exposure to low doses of carcinogenic chemicals to be additive, regardless of the chemicals' mechanisms of toxicity or sites of action (organs of the body). Cancer risk arising from exposure to multiple chemicals in a given exposure medium and pathway is estimated from the following equation (EPA, 1989a):

Eq. 5.3

$$ILCR_p = ILCR_{(chem\,1)} + ILCR_{(chem\,2)} + \cdots ILCR_{(chem\,i)}$$

where:

$ILCR_p$	=	total pathway risk of cancer incidence, calculated
$ILCR(_{chem\ i})$	=	individual chemical cancer risk for the pathway.

The sum of the ILCRs summed across pathways is the total ILCR as shown in the equation below:

Eq. 5.4

$$Total\ ILCR = ILCR_{(p\ 1)} + ILCR_{(p\ 2)} + \cdots ILCR_{(p\ i)}$$

where:

Total ILCR	=	total incremental lifetime cancer risk across all pathways
$ILCR_{pi}$	=	incremental lifetime cancer risks associate with pathway "*i*."

The total ILCR represents all additional cancer risks posed to a given receptor by contact with contaminants in site environmental media.

Total ILCRs in the range of 1E-6 to 1E-4 are regarded as acceptable (EPA, 1990); as mentioned in Section 2.4.1, this range is referred to as the "NCP risk management range." Risks less than this range are regarded as negligible. A target cancer risk criterion of 1E-5 is used by OEPA and will be used in the BHHRA. Use of this 1E-5 criterion represents a departure from the Army's practice of generally using a cancer risk exceeding a value of 1E-4 (the upper end of the NCP risk management range) to trigger remedial action considerations.

5.2 Noncancer Effects of Chemicals

The hazards associated with noncancer effects of chemicals are evaluated by comparing an exposure level or intake with an RfD. The HQ, defined as the ratio of intake to RfD, is estimated as follows (EPA, 1989a):

Eq. 5.5

$$HQ = I\ /\ RfD$$

where:

HQ = hazard quotient (unitless, calculated)
I = intake of chemical averaged over subchronic or chronic exposure period (mg/kg-day)
RfD = reference dose (mg/kg-day).

The I term in Equation 5.4 is equivalent to the "I" or "DAD" terms (intake or dose) in Equations 3.22 through 3.25 and 3.32 when these equations are evaluated for noncancer intakes.

Chemical noncancer hazards are evaluated using chronic RfD values. This approach is different from the probabilistic approach used to evaluate cancer risks. An HQ of 0.01 does not imply a 1-in-100 chance of an adverse effect, but indicates only that the estimated intake is 100 times lower than the RfD. An HQ of unity indicates that the estimated intake equals the RfD. If the HQ is greater than unity, there may be concern for potential adverse health effects.

In the case of simultaneous exposure of a receptor to multiple chemicals, or to a given chemical by multiple pathways, an HI is calculated as the sum of the HQs by the following equation:

Eq. 5.6

$$HI = HQ_1 + HQ_2 + \ldots HQ_i$$

where:

HI = hazard index (unitless, calculated)
HQ_i = hazard quotient for the ith chemical, or for the ith pathway.

A total HI is calculated as the sum of all HI values, including all media and all COPCs, for a given receptor. Calculating a total HI as the sum of HQ values is based on the assumption that the potential for noncancer effects is additive. EPA (1989a), however, acknowledges that the assumption of additivity is probably appropriate only for chemicals that induce adverse effects by the same mechanism (please see Section 4.4). Therefore, if the total HI for a receptor exceeds 1, individual HI values may be calculated for each target organ.

A total target organ HI is calculated by summing the HI values (associated by target organ[s]), across exposure pathways as follows:

Eq. 5.7

$$Total\ Target\ Organ\ HI_a = HI_{p1-a} + HI_{p2-a} + ...HI_{pi-a}$$

where:

Total target organ HI_a = total hazard index for target organ "*a*" (unitless, calculated)
HI_{pi-a} = hazard index for target organ "*a*" via pathway "*i*."

HI values of 1 or less indicate that adverse noncancer health effects associated with that target organ of any individual under the exposure assumptions for that receptor are unlikely. If the total target organ HI exceeds a value of 1, then adverse noncancer health effects concerning that target organ and receptor cannot be regarded as unlikely.

6.0 Uncertainty Analysis

The primary objective of the BHHRA is to characterize and quantify potential human health risks. However, these risks are estimated using incomplete and imperfect information that introduces uncertainties at various stages of the risk assessment process. Uncertainties associated with earlier stages of the risk assessment become magnified when they are concatenated with other uncertainties in the latter stages. Reliance on a simplified numerical presentation of dose rate and risk without consideration of uncertainties, limitations, and assumptions inherent in their derivation can be misleading. For example, the calculated ILCR for a given scenario "A" may be 1E-5 (meets the OEPA risk criterion) and that of scenario "B" may be 5E-5 (exceeds the OEPA risk criterion). However, if the uncertainties associated with scenario "B" span, for instance, orders of magnitude and the ILCR is regarded as biased high, it is not unlikely that scenario "A" actually presents a higher risk of developing cancer.

The chief goal of this analysis is to evaluate uncertainties and present them in context of their potential impact on the interpretation of the risk assessment results and the types of environmental management decisions that may be based on these results. The uncertainty analysis does not exhaustively describe all potential uncertainties but presents those that have the largest implications for the interpretation of the risk assessment results. This analysis also overviews the types and, as applicable, the magnitude of the uncertainties at each stage of the risk assessment. Although the BHHRA will include generic uncertainties that are common to the state of human health risk assessment practice (e.g., additivity of health effects in the risk characterization), overall, the uncertainty analysis will focus on a set of uncertainties that is peculiar to the specific PBOW sites.

6.1 Types of Uncertainty

Uncertainties in risk assessment are categorized into two general types: 1) variability inherent in the (true) heterogeneity of the data set, measurement precision, and measurement accuracy; and 2) uncertainty that arises from data gaps. Estimates of the degree of variability tend to decrease as the sample size increases. This is because larger data sets are less impacted by individual samples/measurements and typically allow for greater accuracy. Uncertainty that arises from data gaps is addressed by applying models and assumptions. Models are applied because they represent a level of understanding to address certain exposure parameters that are impractical or impossible to measure (e.g., COPC concentrations in air that would result from groundwater use that has not yet occurred—or may never occur—at the site). Assumptions represent an educated estimate to address information that is not available (e.g., additivity of carcinogens).

6.2 Sources of Uncertainty

A discussion will be provided that presents an overview of general sources of uncertainty and focuses on those most likely to affect the interpretation of the BHHRA results. The sources of uncertainty may include, but are not limited to, the following:

- Representativeness of samples
- Laboratory procedures and analytical methods
- Sampling methods
- Adequacy of background data set
- Comparisons to background concentrations
- Land use and groundwater use assumptions
- Routes of exposure
- Exposure assessment values
- Exposure models
- Methods of calculating EPCs
- Toxicity values
- Form or isomer of chemical
- Interactions of multiple contaminants.

The PBOW groundwater BHHRA will identify and describe the unique set of uncertainties associated with the site. Special attention may be given to those uncertainties that are thought to have the most significant impact on risk and/or remediation decisions.

7.0 Development of Risk-Based Remediation Criteria

RBRCs are derived to provide support for risk management decisions. Thus, they are developed only for the chemicals of concern (COC) in media that are associated with unacceptable risk that may potentially warrant corrective action. RBRCs are back-calculated from the risk characterization results, which reflect the site-specific concentrations, exposure assumptions, and toxicity assumptions applied in the BHHRA. Consequently, the RBRCs are site-, source medium-, receptor-, and chemical-specific. RBRCs are values based on specific risk (i.e., ILCR=1E-6, 1E-5) or hazard levels (i.e., HQ=0.1, 1). They are intended to indicate a range within which cleanup values may be developed during the FS process, should the medium in question require a remedial action. RBRCs are not intended to serve as final cleanup criteria. Further information such as site-specific conditions, spatial orientation of the contamination, other COCs, other contaminated media, and remedial action objectives should be considered in the development of the final cleanup levels during the FS process.

COCs are preliminarily identified in the BHHRA as site-related chemicals that either exceed a medium-specific applicable or relevant and appropriate requirement or contribute significantly to an unacceptable risk or hazard. Significant contribution to cancer risk is defined as that associated with a COPC (all exposure pathways for a given receptor and medium) which is estimated as having an ILCR of 1E-6 or greater. Significant contribution to noncancer hazard is defined as that associated with a COPC (all exposure pathways for a given receptor and medium) which is estimated as having a target organ-specific HQ of 0.1 or greater. The list of COCs identified in the BHHRA may be revised by the project delivery team during the FS process based on other site-specific considerations.

Separate sets of RBRC values will be derived for each WWTP and AP site at which COCs are identified. As stated above, the RBRCs are back-calculated using the risk characterization results. An RBRC for a COC that is based on cancer effects is derived for a given medium from the following equation:

Eq. 7.1

$$RBRC_{coc-R} = \frac{EPC_{coc}\ TR}{ILCR_{coc-R}}$$

where:

$RBRC_{coc\text{-}R}$	=	remedial goal option for a given COC, receptor and source medium (calculated)
EPC_{coc}	=	exposure point concentration of the COC in the given medium
TR	=	target risk level (1E-6, 1E-5)
$ILCR_{coc\text{-}R}$	=	total incremental lifetime cancer risk for a given COC, receptor, and source medium combination.

An RBRC for the noncancer effects of a COC in a given medium is derived as follows:

Eq. 7.2

$$RBRC_{coc-R} = \frac{EPC_{coc}\ THI}{HQ_{coc-R}}$$

where:

$RBRC_{coc\text{-}R}$	=	remedial goal option for a given COC, receptor and source medium (calculated)
EPC_{coc}	=	exposure point concentration of the COC in the given medium
THI	=	target hazard index (0.1, 1)
$HQ_{coc\text{-}R}$	=	hazard quotient for a given COC, receptor and source medium combination.

Concentration units are not provided in Equations 7.1 and 7.2; the RBRC units will be the same as the concentration units of the EPC. Both cancer-based and noncancer-based RBRCs will be derived for COCs for which both cancer and noncancer-based toxicity values are available.

8.0 Summary and Conclusions

This section will briefly summarize the BHHRA protocol and results and interpret the results, in light of the uncertainty about their estimation, to draw appropriate conclusions regarding risks and hazards to human health.

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TABLES

Table 3-1

Receptor/Exposure Scenarios
Waste Water Treatment Plants 1 and 3; Ash Pits 1 and 3
Plum Brook Ordnance Works, Sandusky, Ohio

Source Medium	Model	Exposure Medium	Exposure Route
Groundskeeper – Current			
Surface Soil	None	Soil	Incidental Ingestion
			Dermal Contact
	Dust Emissions Based on Activity	Ambient Air	Inhalation
	Volatilization from Soil	Ambient Air	Not Quantified[a]
Subsurface Soil	Not Quantified[b]		
Groundwater	Not Quantified[c]		
Sediment	Not Quantified[b]		
Groundskeeper – Future			
Total Soil[d]	None	Soil	Incidental Ingestion
			Dermal Contact
	Dust Emissions Based on Activity	Ambient Air	Inhalation
	Volatilization from Soil	Ambient Air	Inhalation
Groundwater	None	Tap Water	Ingestion
			Dermal Contact
Sediment	Not Quantified[b]		
Indoor Worker – Future[e]			
Surface Soil	None	Soil	Incidental Ingestion
			Dermal Contact[a]
	Dust Emissions; Volatilization	Indoor Air	Not Quantified[a]
Subsurface Soil	Volatilization from Soil	Indoor Air	Inhalation
Surface Water	Not Quantified[c]		
Sediment	Not Quantified[c]		

Table 3-1

Receptor/Exposure Scenarios
Waste Water Treatment Plants 1 and 3; Ash Pits 1 and 3
Plum Brook Ordnance Works, Sandusky, Ohio

Source Medium	Model	Exposure Medium	Exposure Route
Construction Worker – Current/Future			
Total Soil[d]	None	Soil	Incidental Ingestion
			Dermal Contact
	Dust Emissions Based on Activity	Ambient Air	Inhalation
	Volatilization from Soil	Ambient Air	Inhalation
Groundwater	Not Quantified[b]		
Surface Water	None	Surface Water	Dermal Contact
	Volatilization from Water	Ambient Air	Not Quantified[a]
Sediment	None	Sediment	Incidental Ingestion
			Dermal Contact
On-Site Resident – Future			
Total Soil[d]	None	Soil	Incidental Ingestion
			Dermal Contact
	Dust Emissions Based on Wind Erosion	Ambient Air	Inhalation
	Volatilization from Soil	Ambient Air	Inhalation
Subsurface Soil	Volatilization from Soil	Indoor Air	Inhalation
Groundwater	None	Tap Water	Ingestion
			Dermal Contact
	Volatilization from Water	Indoor Air	Inhalation
Surface Water	None	Surface Water	Dermal Contact
	Volatilization from Water	Ambient Air	Not Quantified[a]
Sediment	None	Sediment	Incidental Ingestion
			Dermal Contact

Table 3-1

Receptor/Exposure Scenarios
Waste Water Treatment Plants 1 and 3; Ash Pits 1 and 3
Plum Brook Ordnance Works, Sandusky, Ohio

Source Medium	Model	Exposure Medium	Exposure Route
Hunter – Future			
Surface Soil	None	Soil	Incidental Ingestion
			Dermal Contact
	Dust Emissions, Volatilization	Ambient Air	Not Quantified[a]
	Biouptake	Venison	Venison Consumption
Subsurface Soil	Not Quantified[c]		
Surface Water	Not Quantified[b]		
Sediment	Not Quantified[b]		
Hunter's Child – Future			
Surface Soil	Not Quantified[c]		
	Not Quantified[c]		
	Biouptake	Venison	Venison Consumption
Subsurface Soil	Not Quantified[c]		
Surface Water	Not Quantified[c]		
Sediment	Not Quantified[c]		

[a] Although theoretically complete, this pathway is not quantified as explained in text.
[b] Although contact with this medium is possible, exposure would be sporadic, rather than continuous or predictable. Such exposures do not lend themselves to evaluation under the chronic toxicity paradigm used in a baseline risk assessment.
[c] There is no plausible pathway for exposure.
[d] Total soil represents a mixture of surface and subsurface soil. This is assumed for future scenarios where excavation and regarding is assumed to take place.
[e] Even though the mixing of surface and subsurface soil described in footnote "c" might otherwise be applicable, this receptor was selected primarily to evaluate exposure to indoor air resulting from subsurface soil contamination. Surface soil was used for direct contact exposure to avoid potential "double counting" of contaminants in subsurface soil (refer to Section 3.1.3.2 of text).

Table 3-2

Variables Used to Estimate Potential Chemical Intakes and Contact Rates for Receptors
Waste Water Treatment Plants 1 and 3; Ash Pits 1 and 3
TNT Areas A and C, Plum Brook Ordnance Works, Sandusky, Ohio

Pathway Variable	Grounds-keeper	Construction Worker	On-Site Resident	Indoor Worker	Hunter and Hunter's Child
General Variables Used in All Intake Models					
Body weight (BW), kg	70[a]	70[a]	Child: 15[a] Adult: 70[a]	70[a]	Child: 15[a] Adult: 70[a]
Averaging time, noncancer (AT), days[b]	9125	183	Child: 2190 Adult: 8760	9125	Child: 2190 Adult: 10950
Averaging time, cancer (AT), days[b]	25550	25550	25550	25550	25550
Inhalation of VOCs and Resuspended Dust from Surface Soil, Total Soil or Subsurface Soil					
Fraction exposed to contaminated medium (FI_a), unitless	1[c]	1[c]	1[c]	NA	NA
Inhalation rate (IR_a), m^3/day	20[d]	20[d]	Child: 10[e] Adult: 20[d]	NA	NA
Exposure frequency (EF), days/year	250[a]	250[a]	350[a]	NA	NA
Exposure duration (ED), years	25[a]	0.5[c]	Child: 6[a] Adult: 24[a]	NA	NA
Inhalation of VOCs in Indoor Air from Subsurface Soil					
Fraction exposed to contaminated medium (FI_a), unitless	NA	NA	1[c]	1[c]	NA
Inhalation rate (IR_a), m^3/day	NA	NA	Child: 10[e] Adult: 20[d]	20[d]	NA
Exposure frequency (EF), days/year	NA	NA	350[a]	250[d]	NA
Exposure duration (ED), years	NA	NA	Child: 6[a] Adult: 24[a]	25[a]	NA
Incidental Ingestion of Soil					
Fraction exposed to contaminated medium (FI_{so}), unitless	1[c]	1[c]	0.9[f]	1[c]	1[c]
Soil incidental ingestion rate (IR_{so}), mg/day	100[a]	330[a]	Child: 200[a] Adult: 100[a]	50[a]	Child: NA Adult: 100[a]

Table 3-2

Variables Used to Estimate Potential Chemical Intakes and Contact Rates for Receptors
Waste Water Treatment Plants 1 and 3; Ash Pits 1 and 3
TNT Areas A and C, Plum Brook Ordnance Works, Sandusky, Ohio

Pathway Variable	Grounds-keeper	Construction Worker	On-Site Resident	Indoor Worker	Hunter and Hunter's Child
Exposure frequency (EF), days/year	250[a]	250[a]	350[a]	250[d]	14[d]
Exposure duration (ED), years	25[a]	0.5[c]	Child: 6[a] Adult: 24[a]	25[a]	30[a]
Incidental Ingestion of Sediment					
Fraction exposed to contaminated medium (FI_{sd}), unitless	NA	1[c]	0.1[f]	NA	NA
Sediment incidental ingestion rate (IR_{sd}), mg/day	NA	330[a]	Child: 200[a] Adult: 100[a]	NA	NA
Exposure frequency (EF), days/year	NA	250[a]	350[a]	NA	NA
Exposure duration (ED), years	NA	0.5[c]	Child: 6[a] Adult: 24[a]	NA	NA
Dermal Contact with Soil					
Fraction exposed to contaminated medium (FI_{so}), unitless	1[c]	1[c]	0.9[f]	NA	1[c]
Body surface area exposed to soil (SA_{so}), cm^2	3,300[g]	3,300[g]	Child: 2,800[g] Adult: 5,700[g]	NA	Child: NA Adult: 3,300[c]
Soil-to-skin adherence factor (AF_{so}), mg/cm^2	0.2[g]	0.3[g]	Child: 0.2[g] Adult: 0.07[g]	NA	0.2[c]
Dermal absorption factor (ABS), unitless	csv	csv	csv	NA	csv
Exposure frequency (EF), days/year	250[a]	250[a]	350[a]	NA	14[c]
Exposure duration (ED), years	25[a]	0.5[c]	Child: 6[a] Adult: 24[a]	NA	30[a]

Table 3-2

Variables Used to Estimate Potential Chemical Intakes and Contact Rates for Receptors
Waste Water Treatment Plants 1 and 3; Ash Pits 1 and 3
TNT Areas A and C, Plum Brook Ordnance Works, Sandusky, Ohio

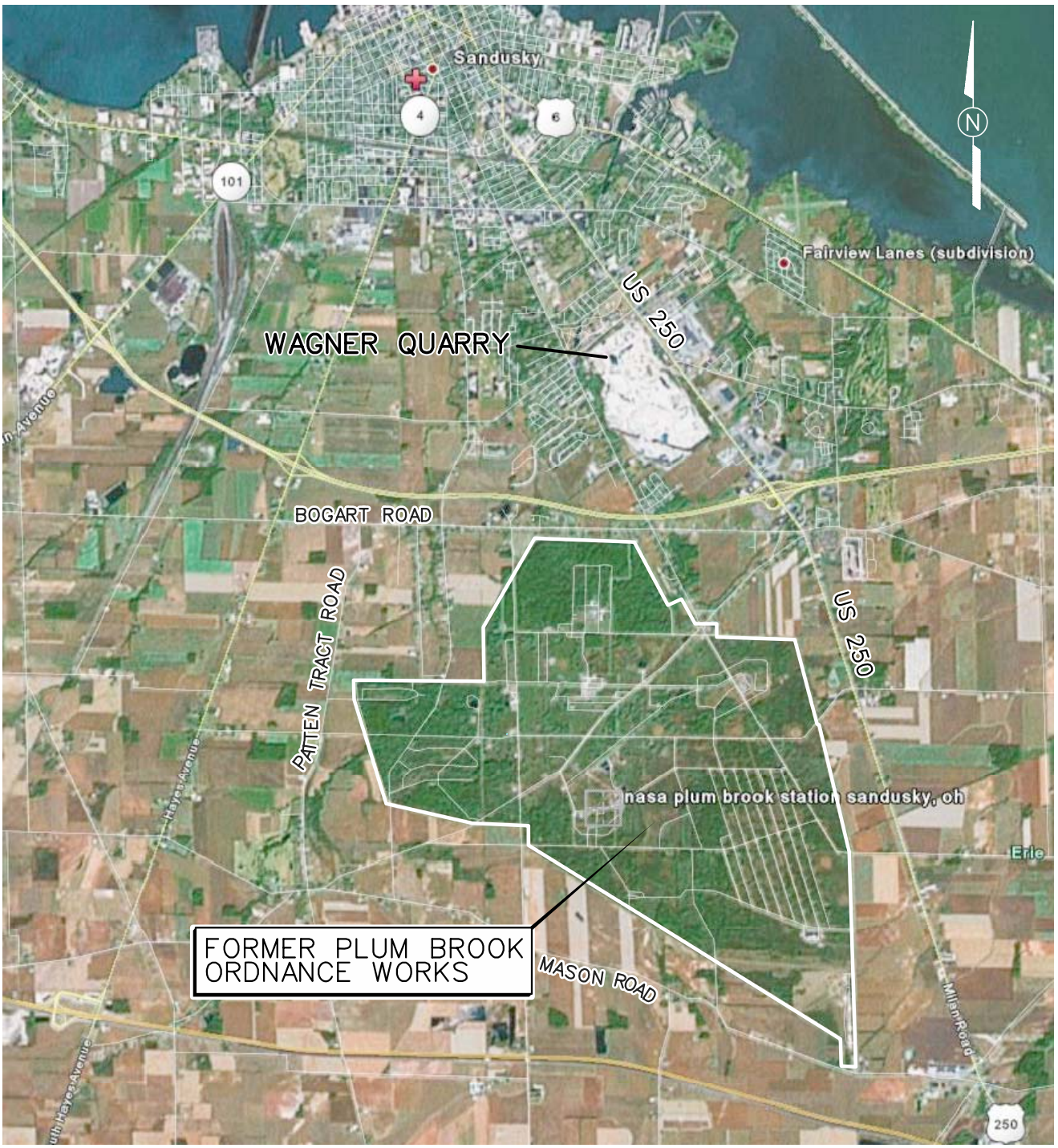
Pathway Variable	Grounds-keeper	Construction Worker	On-Site Resident	Indoor Worker	Hunter and Hunter's Child
Dermal Contact with Sediment					
Fraction exposed to contaminated medium (FI_{sd}), unitless	NA	1[c]	0.1[f]	NA	NA
Body surface area exposed to sediment (SA_{sd}), cm^2	NA	3,300[g]	Child: 2,800[g] Adult: 5,700[g]	NA	NA
Sediment-to-skin adherence factor (AF_{sd}), mg/cm^2	NA	0.3[g]	Child: 0.2[g] Adult: 0.07[g]	NA	NA
Dermal absorption factor (ABS), unitless	NA	csv	csv	NA	NA
Exposure frequency (EF), days/year	NA	250[a]	52[c,f]	NA	NA
Exposure duration (ED), years	NA	0.5[c]	Child: 6[a] Adult: 24[a]	NA	NA
Dermal Contact with Surface Water					
Body surface area exposed to surface water (SA_{sw}), cm^2	NA	3,300[h]	Child: 2,800[h] Adult: 7,000[c]	NA	NA
Permeability coefficient (PC), cm/hour	NA	csv	csv	NA	NA
Exposure time (ET_{sw}), hour/day	NA	4[c]	3[c]	NA	NA
Exposure frequency (EF), days/year	NA	250[a]	52[c]	NA	NA
Exposure duration (ED), years	NA	0.5[c]	Child: 6[a] Adult: 24[a]	NA	NA
Venison Consumption					
Venison ingestion rate (IR_v), kg/day	NA	NA	NA	NA	Child: 0.005[c] Adult: 0.013[c]
Exposure frequency (EF), days/year	NA	NA	NA	NA	350[a]
Exposure duration (ED), years	NA	NA	NA	NA	Child: 6[a] Adult: 30[c]

Table 3-2

Variables Used to Estimate Potential Chemical Intakes and Contact Rates for Receptors
Waste Water Treatment Plants 1 and 3; Ash Pits 1 and 3
TNT Areas A and C, Plum Brook Ordnance Works, Sandusky, Ohio

[a] U.S. Environmental Protection Agency (EPA), 2002, *Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites*, Office of Solid Waste and Emergency Response, Washington, DC, 9355.4-24, December.
[b] For noncancer evaluation, calculated as the product of ED (years) x 365 days/year; for cancer evaluation, calculated as the product of 70 years (assumed human lifetime) x 365 days/year. Source: U.S. Environmental Protection Agency (EPA), 1989a, *Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual (Part A)*, Interim Final, Office of Emergency and Remedial Response, Washington, D.C., EPA/540/1-89/002.
[c] Assumed; see text.
[d] U.S. Environmental Protection Agency (EPA), 1991, *Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual Supplemental Guidance, Standard Default Exposure Factors*, Interim Final, Office of Solid Waste and Emergency Response, OSWER Directive: 9285.603.
[e] U.S. Environmental Protection Agency (EPA), 2004a, *User's Guide and Background Technical Document for Region 9 Preliminary Remediation Goals (PRG) Table*, Region 9, San Francisco, California, October, <http://www.epa.gov/region09/waste/sfund /prg/files/04usersguide.pdf>.
[f] It is assumed that on days when the resident is visiting the ditches and is exposed to sediment that half of the daily exposure via dermal contact and ingestion are associated with ditch sediment (sediment FI=0.5) and half of the exposure is associated with soil (soil FI=0.5). The resident is assumed to be exposed to soil 350 days/year and to sediment 52 days/year. The FI values of 0.1 for sediment and 0.9 for soil are weighted average daily values as described in Section 3.1.3.4 of the text.
[g] U.S. Environmental Protection Agency (EPA), 2004b, *Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual (Part E - Supplemental Guidance for Dermal Risk Assessment)*, Final, Office of Superfund Remediation and Technology Innovation, Washington, D.C., EPA/540/R-99/005, July.
[h] Value for dermal soil exposure (EPA, 2004b) was selected as appropriate for exposure to this medium by this receptor; refer to text for detail.

FIGURES



NOT TO SCALE

FIGURE 1-1
PBOW VICINITY MAP

WWTP1 &3 AND AP1 &3 BASELINE HUMAN HEALTH RISK ASSESSMENT WORK PLAN
FORMER PLUM BROOK ORDNANCE WORKS
NASA PLUM BROOK STATION
SANDUSKY, OHIO

Shaw® Shaw Environmental & Infrastructure, Inc.

STARTING DATE: 05/27/09	DATE LAST REV.:	DRAFT. CHCK. BY:	INITIATOR: T. SIARD	132457_blhhrawp_001.dgn
DRAWN BY: C. BENTLEY	DRAWN BY:	ENGR. CHCK. BY:	PROJ. MGR.: S. DOWNEY	PROJ. NO.: 132457

PBOW_standard.tbl Canon-color-Letter.plt
cbentley 2:37:41 PM
132457_blhhrawp_001.dgn 6/15/2009

LEGEND:

- OVERBURDEN MONITORING WELL LOCATION
- BEDROCK MONITORING WELL LOCATION
- AREA OF CONCERN
- POND
- CREEK, DITCH, CONVEYANCE
- RAILROAD
- ROAD
- FENCE (DENOTES FACILITY BOUNDARY)

NOTES:

1. *SNAKE ROAD BURNING GROUND AREA WAS ALSO REFERRED TO AS "SNAKE ROAD BURN PIT", "SCHEID ROAD BURNING GROUNDS", AND "DISPOSAL AREA THREE".
2. REACTOR FACILITY CONSTRUCTED BY NASA POST WWII AND IS IDENTIFIED FOR LOCATION PURPOSES ONLY.

STARTING DATE	INITIATOR	CHKD	DRAWN	CHKD	PROJECT MNGR
05/27/09	T. SIARD	T. SIARD	C.BENTLEY		S.DOWNEY

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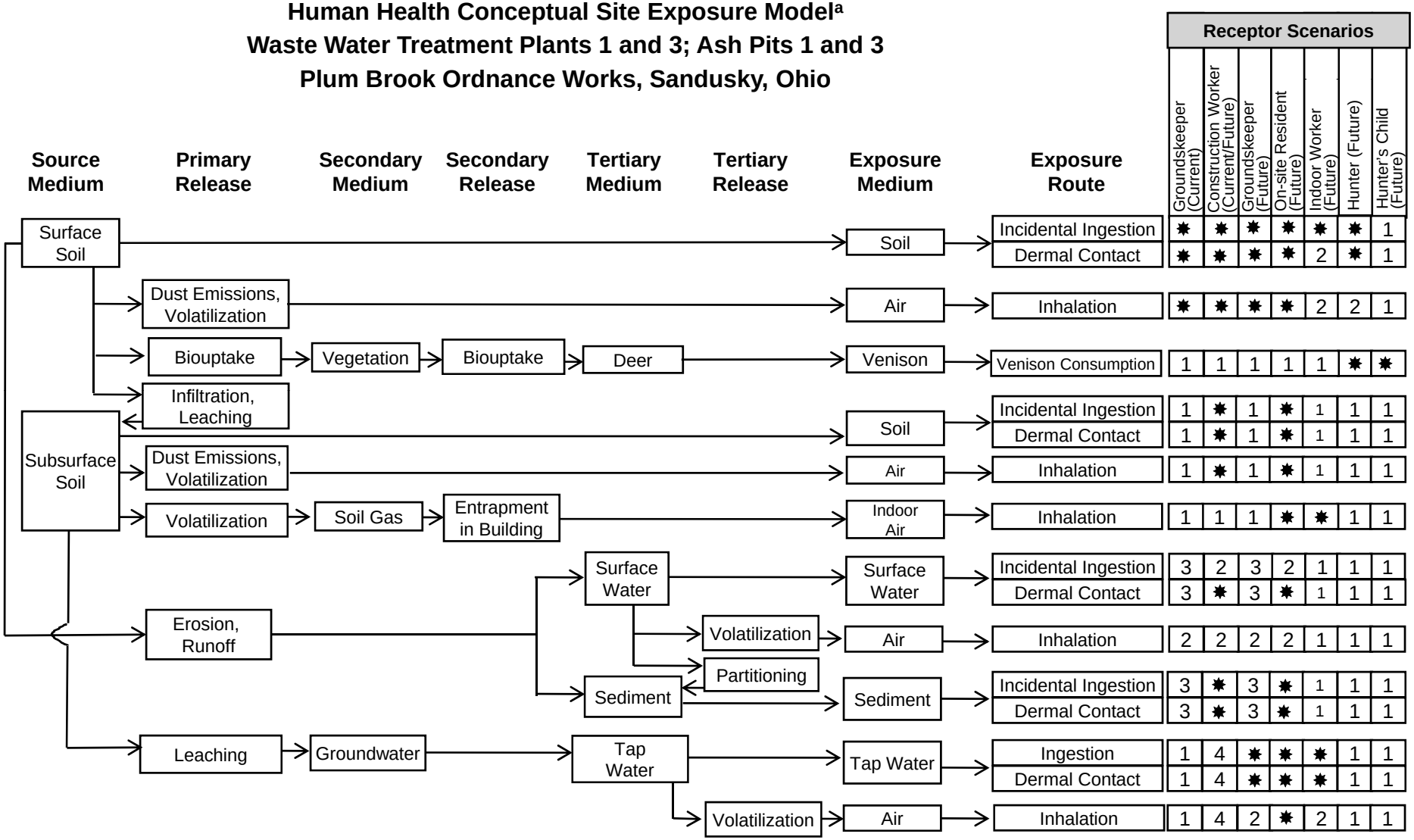
KNOXVILLE, TENNESSEE

WWTP1 &3 AND AP1 &3 BASELINE HUMAN HEALTH RISK ASSESSMENT WORK PLAN
FORMER PLUM BROOK ORDNANCE WORKS
NASA PLUM BROOK STATION
SANDUSKY, OHIO

FIGURE 1-2
LOCATIONS OF AREAS OF CONCERN

AREA	JOB NO.	DRAWING NO.	REV
	132457	132457_BLHHRAWP_002	

Figure 3-1
Human Health Conceptual Site Exposure Model[a]
Waste Water Treatment Plants 1 and 3; Ash Pits 1 and 3
Plum Brook Ordnance Works, Sandusky, Ohio



Exposure Route	Groundskeeper (Current)	Construction Worker (Current/Future)	Groundskeeper (Future)	On-site Resident (Future)	Indoor Worker (Future)	Hunter (Future)	Hunter's Child (Future)
Incidental Ingestion (Surface Soil → Soil)	✸	✸	✸	✸	✸	✸	1
Dermal Contact (Surface Soil → Soil)	✸	✸	✸	✸	2	✸	1
Inhalation (Dust Emissions, Volatilization → Air)	✸	✸	✸	✸	2	2	1
Venison Consumption (Biouptake → Vegetation → Biouptake → Deer → Venison)	1	1	1	1	1	✸	✸
Incidental Ingestion (Subsurface Soil → Soil)	1	✸	1	✸	1	1	1
Dermal Contact (Subsurface Soil → Soil)	1	✸	1	✸	1	1	1
Inhalation (Subsurface Soil → Dust Emissions, Volatilization → Air)	1	✸	1	✸	1	1	1
Inhalation (Subsurface Soil → Volatilization → Soil Gas → Entrapment in Building → Indoor Air)	1	1	1	✸	✸	1	1
Incidental Ingestion (Erosion, Runoff → Surface Water → Surface Water)	3	2	3	2	1	1	1
Dermal Contact (Erosion, Runoff → Surface Water → Surface Water)	3	✸	3	✸	1	1	1
Inhalation (Erosion, Runoff → Surface Water → Volatilization → Air)	2	2	2	2	1	1	1
Incidental Ingestion (Erosion, Runoff → Surface Water → Partitioning → Sediment → Sediment)	3	✸	3	✸	1	1	1
Dermal Contact (Erosion, Runoff → Surface Water → Partitioning → Sediment → Sediment)	3	✸	3	✸	1	1	1
Ingestion (Leaching → Groundwater → Tap Water → Tap Water)	1	4	✸	✸	✸	1	1
Dermal Contact (Leaching → Groundwater → Tap Water → Tap Water)	1	4	✸	✸	✸	1	1
Inhalation (Leaching → Groundwater → Tap Water → Volatilization → Air)	1	4	2	✸	2	1	1

a = All groundwater evaluation is deferred to a site-wide groundwater investigation to be conducted in the future.
✸ = Complete exposure route quantified in the risk assessment.
1 = There is no plausible pathway for exposure to this medium.
2 = Although theoretically complete, this pathway is not quantified as explained in text.
3 = Contact with this medium, although plausible, is not part of this receptor's normal or expected activities; therefore contact would be sporadic and is not quantified.
4 = For current use there is no plausible exposure pathway. For future use, the pathway is potentially complete, but is not quantified as explained in the text.

KN9/PBOW/WWTP/P1-3/WP/BHHRA/Fig3-1.ppt/5/27/09

Screening Level Ecological Risk Assessment
Work Plan
Waste Water Treatment Plants 1 and 3
and Ash Pits 1 and 3

Former Plum Brook Ordnance Works, Sandusky, Ohio

Prepared for:

Plum Brook Ordnance Works
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Table of Contents

Page

List of Figures iii
List of Acronyms iv
1.0 Introduction 1-1
1.1 Facility Description 1-2
1.2 Background 1-2
1.2.1 Waste Water Treatment Plants 1-2
1.2.2.1 WWTP1 1-3
1.2.1.2 WWTP3 1-3
1.2.2 Ash Pit Areas 1-4
1.2.2.1 AP1 1-4
1.2.2.2 AP3 1-5
1.3 Scope and Objectives 1-6
2.0 Problem Formation 2-1
2.1 Ecological Site Description 2-1
2.2 Pre-Assessment Reconnaissance (Biota Checklist) 2-2
2.3 Documentation of Potential Receptors of Special Concern and Critical Habitat 2-3
2.4 Significant Ecological Threats 2-4
2.5 Review, Evaluation, and Presentation of Analytical Data 2-4
2.6 Selection of Chemicals of Potential Ecological Concern 2-4
2.6.1 Data Organization 2-4
2.6.2 Descriptive Statistical Calculations 2-5
2.6.3 Frequency of Detection 2-6
2.6.4 Background Evaluation 2-6
2.6.5 Essential Nutrients 2-7
2.6.6 Comparison to Ecological Screening Values 2-8
2.6.7 Summary of COPEC Selection 2-9
2.7 Selection of Assessment Receptors 2-10
2.8 Ecological Endpoint (Assessment and Measurement) Identification 2-11
2.9 Ecological Site Conceptual Model 2-12
3.0 Exposure Characterization 3-1

Table of Contents *(continued)*

Page

3.1 Exposure Analysis 3-1
3.2 Exposure Characterization Summary 3-6
4.0 Ecological Effects Characterization 4-1
4.1 Selection of Literature Benchmark Values 4-1
4.2 Development of Reference Toxicity Values 4-1
5.0 Risk Characterization 5-1
5.1 Risk Estimation 5-1
5.2 Uncertainty Analysis 5-2
5.3 Risk Description 5-3
6.0 Risk Summary and Conclusions 6-1
7.0 References 7-1

List of Figures

Figure	*Title*
1-1	Plum Brook Ordnance Works Location Map
4-1	Procedural Flow Chart for Deriving Toxicity Reference Values (TRV) from Class-Specific Toxicity Data

List of Acronyms

AP	Ash Pit
ARP	assessment receptor profile
BAF	bioaccumulation factor
bgs	below ground surface
BSC	background screening concentration
BTAG	Biological Technical Assistance Group
COPEC	chemical(s) of potential ecological concern
D&M	Dames and Moore, Inc.
DERP	Defense Environmental Restoration Program
DNT	dinitrotoluene
DOD	U.S. Department of Defense
EPA	U.S. Environmental Protection Agency
EPC	exposure point concentration
ESCM	ecological site conceptual model
ESV	ecological screening value
FCM	food chain multiplier
FUDS	Formerly Used Defense Sites
HQ	hazard quotient
IT	IT Corporation
K_{ow}	octanol-water partition coefficient
LOAEL	lowest observed adverse effect level
MDC	maximum detected concentration
NASA	National Aeronautics and Space Administration
NOAEL	no observed adverse effect level
OEPA	Ohio Environmental Protection Agency
PBOW	Plum Brook Ordnance Works
Shaw	Shaw Environmental, Inc.
SLERA	screening level ecological risk assessment
TNT	trinitrotoluene
TNTA	TNT Area A
TNTB	TNT Area B
TRV	toxicity reference value
UCL	upper confidence limit

List of Acronyms

USACE	U.S. Army Corps of Engineers
WRS	Wilcoxon Rank Sum
WWTP	Waste Water Treatment Plant

1.0 Introduction

Chemical contamination related to former U.S. Department of Defense (DOD) activities has been documented at the former Plum Brook Ordnance Works (PBOW) located near Sandusky, Ohio (U.S. Army Corps of Engineers [USACE], 2000a, b). PBOW operated from 1941 to 1945 as a manufacturing plant for 2,4,6-trinitrotoluene (TNT), dinitrotoluenes (DNT), and pentolite. Some of the areas used by DOD were decontaminated in the 1950s and 1960s; other areas have been decommissioned but not decontaminated. The site is currently owned by the National Aeronautics and Space Administration (NASA) and is operated as the Plum Brook Station of the John Glenn Research Center, which is headquartered in Cleveland, Ohio. In 1978 NASA declared approximately 2,152 acres of land as excess (IT Corporation [IT], 1997). The Perkins Township Board of Education acquired 46 acres of excess for use as a bus transportation center. The Ohio National Guard has an agreement with the U.S. Army's General Services Administration to use 604 acres of the facility. The areas surrounding PBOW are predominantly agricultural and residential. The facility is currently surrounded by a chain-link fence, and the perimeter is regularly patrolled. Access by authorized personnel is limited to established checkpoints. Public access is restricted, except during the annual deer hunting season.

This screening level ecological risk assessment (SLERA) work plan presents the protocol for evaluating the potential for adverse effects posed to ecological receptors from suspected hazardous substance releases at Waste Water Treatment Plants (WWTP) 1 and 3 and Ash Pits (AP) 1 and 3 at the former PBOW. This work plan is consistent with U.S. Environmental Protection Agency (EPA) and Ohio Environmental Protection Agency (OEPA) – Division of Emergency and Remedial Response (OEPA, 2008) guidance and with the procedures previously established in the Baseline Ecological Risk Assessment Work Plans, TNT Areas A and C (IT, 2001).

This work is being conducted by Shaw Environmental, Inc. (Shaw) for the USACE under the Defense Environmental Restoration Program (DERP) - Formerly Used Defense Sites (FUDS) and managed by the USACE Huntington District and technically overseen by the USACE Nashville District.

1.1 Facility Description

PBOW is located approximately 4 miles south of Sandusky, Ohio, and 59 miles west of Cleveland (Figure 1-1). Although located primarily in Perkins and Oxford Townships, the eastern edge of the facility extends into Huron and Milan Townships. PBOW is bounded on the north by Bogart Road, on the south by Mason Road, on the west by Patten Tract Road, and on the east by U.S. Highway 250. The area surrounding PBOW is mostly agricultural and residential. The facility is currently surrounded by a chain-link fence, and the perimeter is regularly patrolled. Access by authorized personnel is limited to established checkpoints. Public access is restricted, except during the annual deer hunting season.

1.2 Background

The 9,009-acre PBOW site was built in early 1941 as a manufacturing plant for TNT, DNT, and pentolite. Production of explosives began in December 1941 and continued until 1945. It is estimated that more than one billion pounds of explosives were manufactured during the four-year operating period.

NASA acquired PBOW in 1963 and presently utilizes about 6,400 acres for conducting space research. The site is operated by NASA as the Plum Brook Station of the John Glenn Research Center, which is headquartered in Cleveland, Ohio. In 1978 NASA declared approximately 2,152 acres of land as excess (IT, 1997). The Perkins Township Board of Education acquired 46 acres of the excess property for use as a bus transportation center. The Ohio National Guard has an agreement with the U.S. General Services Administration to use 604 acres of the facility.

1.2.1 Waste Water Treatment Plants

Three WWTPs, WWTP1, WWTP2, and WWTP3 at PBOW, received waste water from TNT manufacturing operations. WWTP1 is located nearest to TNT Area A (TNTA) and received waste water from the settling basins located at TNTA and from TNT Area B (TNTB). WWTP2 is located northeast of the West Area Red Water Pond, east of Acid Area No. 2, and north of TNT Area C and received waste water from TNT Area C. WWTP3 is located between WWTP1 and WWTP2 and northwest of TNTB (see Figure 1-2 in the human health work plan). The purpose of the plants was to reduce the volume of waste water discharged from each of the manufacturing areas to the West Area Red Water Pond and Pentolite Road Red Water Pond. The waste water that was received from the TNT manufacturing areas consisted of spent sulfuric and nitric acids and red water from the TNT purification process. Chemicals in the waste

streams included sodium salts of sulfite, sulfate, nitrite, nitrate, sulfonates of unwanted TNT isomers, trinitrobenzoic acid, trinirobenzaldehyde, trinitrobenzyl alcohol, nitrotoluenes, and DNTs (Dames and Moore, Inc. [D&M], 1997).

Each of the WWTPs consisted of a neutral storage tank, an equalization tank, a condensate storage tank, an evaporator building, an incinerator, and a thick liquid storage tank. WWTP1 also included a caustic storage tank and a raw waste storage tank. The plants received waste water from the TNT manufacturing settling basins and neutralized the slurry through a chemical depuration process. The liquid was discharged to open ditches or ponds surrounding the facilities or thickened by evaporation. The thickened liquid was then incinerated. The incinerators were located to the east of the storage tanks at WWTP1 and to the north of the storage tanks at WWTP3. Ash from the incinerator was disposed of in nearby ash pits associated with the WWTPs (USACE, 2000a). The historical processes are discussed further in the following subsections.

1.2.1.1 WWTP1

The former WWTP1 is located in the northern midsection of PBOW, approximately 200 feet northeast of the intersection of Maintenance Road and Taylor Road (see Figure 1-2 in the human health work plan). It occupies approximately 2 acres of land. A former building located approximately 200 feet northeast of WWTP1 is still present and is currently used by NASA.

Limited investigation has been conducted at WWTP1. Historical drawings were reviewed by D&M (1995), and it was determined that the raw waste storage tank located at the former WWTP1 received waste from the settling basins located at manufacturing areas TNTA and TNTB.

During a field reconnaissance in March 1999, bare areas and the remains of concrete foundations were reported to have been observed where the former buildings and storage tanks were once present (USACE, 2000a).

1.2.1.2 WWTP3

Former WWTP3 is located in the western midsection of PBOW, approximately 650 feet southwest of the intersection of Maintenance Road and Ransom Road and 450 feet southeast of the patrol road. It occupies approximately 2 acres of land (Figure 1-2 in the human health work plan). The neutralized waste storage tank of WWTP3 had been used by NASA as the K-Site

control building. The K-Site itself was operated in the former Power House 3 Building (USACE, 2000a). The neutralized waste water storage tank of former WWTP3 had been used by NASA as the control building (USACE, 2000a). D&M (1995) conducted a review of records and found that a bermed, square depression north of WWTP3 and south of Maintenance Road may have been used as an ash settling basin.

1.2.2 Ash Pit Areas

As noted above, PBOW was built in early 1941 and manufactured 2,4,6-TNT, DNT, and pentolite until 1945. Three power stations, Power House 1, Power House 2, and Power House 3, were constructed and utilized to support the TNT manufacturing process. Each power station consisted of a main powerhouse, a coal storage area, and an aboveground fuel storage tank. The fuel storage tank was surrounded by a berm to contain any potential spills or leaks. Each power house building consisted of a boiler house, compressor room, electrical room, filter room, and locker room. The buildings also contained two to four large coal-burning boilers, a turboelectric generator, a feed water treatment system, and several steam-driven or electric air compressors. The generated steam was used for space heating, driving compressors, and generating electrical power. Coal ash generated from each of the boilers in the powerhouse was collected in pits. Water was added to the ash, producing a slurry that flowed through a sluice trench to an ash sump located at the end of each power house. From the ash sump, the ash slurry traveled through a pipeline to a nearby surface water/ash impoundment (AP) (USACE, 2000b). Figure 1-2 in the human health work plan shows the location of the three AP areas on PBOW property.

Based on topographical quadrangles (dated 1959 and 1969), aerial photographs, and a visual site survey conducted in 1999, the AP areas are noted to have essentially remained unchanged. An exception to this is that AP1 has become overgrown with thick vegetation.

1.2.2.1 AP1

Former AP1 is located in the central part of PBOW, approximately 1,095 feet west of the intersection of Maintenance Road and Taylor Road and approximately 50 feet south of Maintenance Road (see Figure 1-2 in the human health work plan).

In June 1999, USACE conducted a limited site investigation of AP1 (USACE, 2000b). AP1 was noted to be overgrown with thick vegetation and a thicket of small trees ranging from 3 to 8 feet tall. During a field reconnaissance by Shaw and USACE personnel in April 2009, the trees (primarily gray dogwood [*Cornus racemosa*]) present in AP1 were approximately 15 feet in

height. Personnel were informed by NASA that an underground high-voltage utility line bisected AP1 in a north-south direction and an underground telephone line was present approximately 5 feet south and parallel to Maintenance Road. USACE also noted a culvert under Maintenance Road and an associated drainage ditch approximately 5 feet due north of AP1. A larger drainage ditch located northwest of AP1 contained water that reportedly flowed in a northeast direction. During site visits in October 2008 and April 2009, USACE and Shaw personnel observed water in this ditch, but it did not appear to be flowing. Standing water was observed in the far north portion of AP1 during April 2009 reconnaissance, especially in ruts made by vehicles and other equipment along Maintenance Road. This standing water was not observed during site visits in September or October 2008. A culvert receives runoff from Maintenance Road, and possibly from AP1, and drains to the north under Maintenance Road.

1.2.2.2 AP3

AP3 is located approximately 700 feet southwest of the intersection of Maintenance and Ransom Roads and is west of the present NASA K-Site control building (former Power House 3) (see Figure 1-2 in the human health work plan). Operations at the K-Site were officially abandoned in 2007; however, it is possible that the facility may be used for test programs in the future. Abandoned railroad tracks running in a north-south direction are immediately east of AP3. The pit is partially surrounded by thick vegetation, with mature and smaller trees. The original surface water impoundment was observed holding water supplied by NASA's K-Site Test Facility in 1999 (USACE, 2000b). Water had been discharged from the former pond by means of an east-west-trending drainage ditch that eventually discharges into Pipe Creek (USACE, 2000b).

Since the K-Site ceased operations, apparently the major source of water to Ash Pit 3 has been eliminated. A review of aerial photographs indicates a lack of open water at AP3 in recent history. NASA (2008) personnel communicated that no water was present in AP3 on July 22, 2008. This was also evident during a September 2008 site visit by USACE and Shaw personnel, although a small ponded area was observed after a rain event during an October 2008 site visit. However, much of Ash Pit 3 was covered by up to approximately 1 foot of water during an April 2009 reconnaissance. This 2009 reconnaissance took place after a precipitation event during a wetter than usual spring.

1.3 Scope and Objectives

A SLERA will be performed to provide an estimate of the potential for adverse ecological effects associated with suspected hazardous substance releases at WWTPs 1 and 3 and APs 1 and 3. The results of the SLERA will contribute to the overall characterization of the site and may be used to determine the need for additional investigations or to develop, evaluate, and select appropriate remedial alternatives. The SLERA will be performed following the general guidelines of the *Tri-Service Procedural Guidelines for Ecological Risk Assessments* (Wentsel, et al., 1996), as well as the *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (EPA, 1997), *Region 5 Biological Technical Assistance Group (BTAG) Ecological Risk Assessment Guidance Bulletin No. 1* (EPA, 1996), and *Guidance for Conducting Ecological Risk Assessments* (OEPA, 2008). The SLERA fits into Steps 1 and 2 of the ecological risk assessment guidance for Superfund process (EPA, 1997), and Level I through a maximum of Level III evaluation using the OEPA (2008) process.

The primary objective of the SLERA is to evaluate the potential for adverse ecological effects to ecological receptors from suspected hazardous substance releases. This objective will be met by characterizing the ecological communities in the vicinity of the site, determining the particular hazardous substances present, identifying pathways for receptor exposure, and estimating the magnitude of the likelihood of potential adverse effects to identified receptors. The SLERA will address the potential for adverse effects to the vegetation, wildlife, aquatic life (sediment-dwelling organisms), threatened and endangered species, and wetlands or other sensitive habitats associated with the site.

Concentrations of chemicals measured in relevant environmental media will be used to perform a SLERA, including problem formulation (Chapter 2.0); exposure characterization (Chapter 3.0); ecological effects characterization (Chapter 4.0); risk characterization (Chapter 5.0); and summary and conclusions (Chapter 6.0). These subtasks are described in greater detail in the following sections.

The chemicals of potential ecological concern (COPEC), the ecosystems and receptors at risk, the ecotoxicity of the contaminants known or suspected to be present, and observed or anticipated ecological effects will be evaluated. This evaluation will be conducted in two steps: (1) a screening assessment step and (2) a predictive assessment step. Ecological endpoints to be addressed in both steps will be identified. The results and conclusions of the screening assessment will determine whether a predictive assessment is needed. The criteria by which the

need for a predictive assessment is measured will be formalized as null hypotheses to be accepted (in which case a predictive assessment is not needed) or rejected (in which case a predictive assessment is needed).

2.0 Problem Formulation

The screening assessment null hypotheses are stated as follows:

- Potential for adverse ecological effects to ecological entities at the sites is minimal or nonexistent due to the lack of viable habitat for potential ecological receptors.
- Potential for adverse ecological effects to ecological entities at the sites is minimal or nonexistent due to the lack of potential ecological receptors.
- Potential for adverse ecological effects to ecological entities at the sites is minimal or nonexistent due to the lack of potential exposure pathways.
- Potential for adverse ecological effects to ecological entities at the sites is minimal or nonexistent due to the lack of potential chemical stressors.

Any determination of a lack of viable habitat or a lack of potential receptors will be qualified with a statement addressing whether or not such absence is due to previous or ongoing site activities. If one or more of these null hypotheses are accepted, a predictive assessment is not triggered. All four null hypotheses must be rejected for a predictive assessment to be triggered. The first three null hypotheses are tested with the results of the ecological site description (Section 2.1), the pre-assessment reconnaissance (Section 2.2), the documentation of potential receptors of special concern and critical habitats (Section 2.3), and the determination of significant ecological threats (Section 2.4). The fourth null hypothesis will be tested with the results of COPEC selection (Sections 2.5 and 2.6).

If a predictive assessment is triggered, terrestrial and aquatic ecological conceptual site models will be developed, as appropriate, and additional problem formulation tasks will be performed as described in Sections 2.7 through 2.9.

2.1 Ecological Site Description

The site will be described in sufficient detail to ensure that technical specialists that review the SLERA can be oriented to the site. This information will be assembled from existing sources. Natural resource personnel (e.g., federal or state officials) will be contacted to obtain any relevant data or useful ecological information. A pre-assessment reconnaissance/ecological survey will be performed to validate the findings, as described in Section 2.2.

2.2 Pre-Assessment Reconnaissance (Biota Checklist)

Shaw will perform a site reconnaissance for the purpose of collecting qualitative information on the type, quality, and location of biological resources at WWTPs 1 and 3 and APs 1 and 3. This will be achieved as follows:

- Dominant plant species will be identified by a qualified botanist, and plant communities will be defined based on dominant species observed.

- Observations of fauna will be made by a qualified biologist or ecologist. Mammals will be identified by tracks, scat, burrows, and sightings. Bird, reptile, and amphibian identifications will be made by sightings. Fish and aquatic macroinvertebrates will be collected for identification as necessary, depending on characteristics of the sites.

- Areas will be examined for vegetative stress. Stress may be exhibited by stunted growth, poor foliage growth, tissue discoloration, and a loss of leaf coverage. Due to the seasonal component of this evaluation, the survey will be performed during mid-autumn, as the schedule permits.

The purpose of these activities will be to select representative receptors, refine exposure scenarios for the risk assessment, and identify protected species or habitats of special concern in the study area.

The site reconnaissance will be performed by Shaw biologists or ecologists. Prior to arrival at the sites, reconnaissance personnel will obtain relevant information on the sites, including topographic maps; township, county, or other appropriate maps; and location of potential ecological units such as streams, creeks, ponds, grasslands, forest, and wetlands on or near the sites. Additionally, the *Biological Inventory of Plum Brook Station, 1994* (NASA, 1995), which identifies and shows the locations of threatened and endangered species at PBOW, as well as results of extensive wildlife surveys, will be reviewed. Reconnaissance personnel will complete a checklist similar to that on EPA's checklist for ecological assessment/sampling (EPA, 1997) and OEPA's ecological risk assessment guidance (OEPA, 2008). The location of known or potential contaminant sources affecting the sites and the probable gradient of the pathway by which contaminants may be released from the sites to the surrounding environment will be identified. Reconnaissance personnel will use the site visit to evaluate the sites for more subtle clues of potential effects from contaminant release. Reconnaissance personnel will determine the designation of any waters potentially impacted by contaminant migration.

Ecological characterization of the study area will be based on a compilation of existing ecological information and site reconnaissance activities. Methods used to characterize ecological resources will include a site walkover for the identification of existing wildlife and vegetative communities; interviews with local, state, and PBOW resource personnel; and a review of environmental data obtained from various sources (e.g., Nature Conservancy, U.S. Fish and Wildlife Service). A photographic record will be made during the site reconnaissance. Information will be obtained on the presence of state-listed and federal-listed, threatened, and endangered species; species of special concern; and wildlife and fisheries resources. A botanist will search for threatened and endangered plant species. A checklist of biological species present at the sites will be developed using existing site investigation reports, environmental data sources mentioned previously, and information gathered during the site reconnaissance. Information on unique and special-concern habitats, preserves, wildlife refuge parks, and natural areas within the general vicinity will also be obtained.

The methods used to characterize natural resources will focus on aquatic and terrestrial resources at the sites and within the immediate vicinity. If not already in existence, general habitat maps will be prepared showing the types and extent of biological communities present within the immediate vicinity of the sites. These maps will be based on information collected during the site reconnaissance previously discussed.

A general ecological reconnaissance of WWTPs 1 and 3 and APs 1 and 3 was performed in October, 2008 by Shaw ecologists and USACE personnel. A fall vegetation survey was conducted at this time, as well. In late April, 2009, a fauna survey was performed at these sites by Shaw ecologists. A follow-up fauna and vegetation survey was performed early June 2009.

2.3 Documentation of Potential Receptors of Special Concern and Critical Habitat

A determination will be made as to whether the sites have designated wetlands or critical or sensitive habitats for threatened or endangered species. This will be performed, in part, by reviewing National Wetland Inventory Maps and threatened and endangered species information requested from the Ohio Department of Natural Resources Division of Natural Areas and Preserves. The site reconnaissance will not include wetlands delineation activities.

2.4 Significant Ecological Threats

A determination will be made as to whether significant ecological threats exist and whether these threats are related to chemical contamination caused by DOD activities. The initial screening of

whether significant threats exist will be based on the qualitative absence of plant or animal life in areas expected to support these ecological components.

2.5 Review, Evaluation, and Presentation of Analytical Data

Chemical analytical data, as well as all previous and ongoing investigations, will be reviewed and evaluated for quality, usefulness, and uncertainty. Data identified as being of acceptable quality for use in the SLERA will be summarized in a manner that presents the pertinent information to be applied in the SLERA. Any data rejected during the data evaluation as a result of the data evaluation ("R"-qualified data) will be identified along with the rejection rationale. Only validated data are proposed for use in the SLERA.

2.6 Selection of Chemicals of Potential Ecological Concern

The selection of COPECs will identify a subset of site-related chemicals to be carried through the risk assessment. The data will be segregated for both WWTPs and APs, and presented in tables in the SLERA report. Screening criteria include analytical detection limit, frequency of detection greater than 5 percent, comparison of inorganic constituent concentrations to naturally occurring background concentrations, role as ecologically essential nutrient concentrations, and comparison of site concentrations to ecologically relevant screening criteria. The COPEC selection process is described in more detail in the following subsections.

2.6.1 Data Organization

The data for each chemical will be sorted by medium. For ecological impacts, soil from 0 to 6 feet below ground surface (bgs) will be considered. Although OEPA has recommended that only soils from 0 to 2 feet bgs be used in the SLERA, OEPA has agreed to the 0 to 6 feet interval, in order to maintain consistency with previous SLERAs performed for the Red Water Ponds and TNTB at PBOW. Chemicals that are not detected at least once in a medium will not be included in the risk assessment. Available background data will be determined for each medium. Potential sources of background information will include data from previous and current investigations, as well as monitoring wells in areas unaffected by site activities.

The analytical data may have qualifiers from the analytical laboratory quality control or from the data validation process that reflect the level of confidence in the data. Some of the more common qualifiers and their meanings are as follows (EPA, 1989):

- U - Chemical was analyzed for but not detected; the associated value is the sample quantitation limit.

- J - Value is estimated, probably below the contract-required quantitation limit.
- R - Quality control indicates that the data are unusable (chemical may or may not be present).
- B - Concentration of chemical in sample is not sufficiently higher than concentration in the blank (using 5X, 10X rule).

"J" qualified data are used in the risk assessment; "R" and "B" qualified data are not. The handling of "U" qualified data (nondetects) is described in the following sections of this work plan.

2.6.2 Descriptive Statistical Calculations

Because of the uncertainty associated with characterizing contamination in environmental media, both the mean and the 95 percent upper confidence limit (UCL) of the mean are usually estimated for each chemical in each medium of interest. The EPA ProUCL software (Version 4.00.04 [EPA, 2009a,b,c]) will be used to estimate UCLs for the data sets of all environmental media represented by at least five samples. If the data set consists of fewer than five data points, the maximum detected concentration (MDC) will be selected as the exposure point concentration (EPC). The method detection limit will be used as the ProUCL input concentration for nondetects. Nondetects with method detection limits greater than the MDCs will not be included in the data set used to calculate the EPC (EPA, 1989), as such values make distribution testing impossible for ProUCL (EPA, 2009c).

ProUCL generates a variety of UCL estimates for each data set. Generally, the results of one or two (sometimes more) of the UCL estimates are recommended. This recommendation is based on a variety of factors, including the distribution (i.e., normal, lognormal, gamma, or not discernable) that provides the best fit, number of nondetects, size of the data set, and skewness. In general, the UCL recommended by ProUCL will be selected as the EPC. Occasionally, ProUCL will recommend the 97.5 or 99 percent UCL on the arithmetic mean estimated by the Chebyshev method. In these cases, the 95 percent UCL estimated by the Chebyshev method is selected as the EPC because this is more consistent with the intent of the reasonable maximum exposure paradigm as defined by EPA (1989; 2002).

Analytical data from field duplicates will be joined with parent sample results to yield one result for use in the generation of mean and UCL concentrations, as follows:

- Use average of field duplicate and parent sample if both are positive detections, or if both are nondetects.

- Use detected value if one sample is a positive detection and the other is a nondetect.

The UCL generated by ProUCL or the MDC, whichever is smaller, will be selected as the EPC and is understood to represent a conservative estimate of average for use in the risk assessment. Unusually high detected values are included in the calculation of the UCL concentration. Inclusion of these high values increases the statistical variability and the overall conservativeness of the risk estimate.

2.6.3 Frequency of Detection

Chemicals that are detected infrequently may be artifacts in the data that may not reflect site-related activity or disposal practices. These chemicals will not be included in the risk evaluation. Generally, chemicals that are detected only at low concentrations in less than 5 percent of the samples from a given medium are dropped from further consideration, unless their presence is expected based on historical information about the site (such as nitroaromatics in the present case). Chemicals detected infrequently at high concentrations may identify the existence of "hot spots" and will be retained in the evaluation, unless other information exists to suggest that their presence is unlikely to be related to site activities.

2.6.4 Background Evaluation

Chemical concentrations will be compared to site-specific background concentrations (see next paragraph for details) as an indication of whether a chemical is present from site-related activity or as natural background. This comparison is generally valid for inorganic chemicals, but not for organic chemicals, because inorganic chemicals are naturally occurring and most organic chemicals are not. Statistical techniques are used as tools to aid the exercise of professional judgment in resolving site-related issues for metals, because metals are naturally present in most environmental media. The statistical techniques generally involve comparing the site data with background data. Background data are only available for soil at PBOW. Background data do not exist for surface water or sediment; therefore, a statistical background evaluation for these media cannot be performed.

The first statistical technique used for the background screen is the comparison of the MDC of

the site data set to the PBOW background screening concentration (BSC). The background data set and derivation of soil BSCs for all PBOW soil investigations is described in IT (1998). The background soil samples were collected from near the property boundary, away from any potential source areas. BSCs were calculated for use at PBOW based on concentrations found in these background soil samples. Each BSC is either the MDC of the concentrations found in these background soil samples or the calculated 95th percent upper tolerance limit of the background data set, whichever value is lower (Shaw, 2005). The upper tolerance limit is the concentration, with a probability of 0.95 (or a confidence of 95 percent), that will capture (or cover) 95 percent of background samples if a large number of samples were taken. Chemicals with MDCs less than the BSC are eliminated from further consideration. If the MDC exceeds the BSC, the chemical is retained as a COPEC, or a different statistical analysis may be performed to determine if the background data and the site data are drawn from the same population. The Wilcoxon Rank Sum (WRS) test is used for this purpose.

The WRS test (also known as the Mann-Whitney U test) is described in Appendix M of Shaw (2005). WRS testing is performed for inorganic chemicals whose MDCs exceed their respective BSCs. Site data sets are interpreted as being significantly different from PBOW background if the associated p-level is less than 0.05. WRS statistical output and box and whisker plots of the various inorganic COPEC data sets will be appended to the SLERA for each inorganic data set evaluated against the site background data set. Analytes shown by the WRS results to exceed background (or for which the WRS testing was not run) are assumed to be site-related and retained as COPECs, unless a qualitative chemical-specific explanation is presented in the uncertainties analysis as to why the analyte should not be regarded as site related. Analytes shown by the WRS results to be drawn from the same population as the background samples will be assumed to be naturally occurring and will not be retained as COPECs.

2.6.5 Essential Nutrients

Evaluating essential nutrients is a special form of risk-based screening applied to certain ubiquitous elements that are generally considered to be required nutrients. Essential nutrients such as calcium, iron, magnesium, potassium, and sodium are usually eliminated as COPECs because they are generally considered to be innocuous in environmental media. Other essential nutrients, including chloride, iodine, and phosphorus, may be eliminated as COPECs, provided that their presence in a particular medium is shown to be unlikely to cause adverse effects to biological health.

2.6.6 Comparison to Ecological Screening Values

A comparison will be made between MDCs of chemicals in media and ecological screening values (ESV) for ecological endpoints following recommendations received from OEPA and as discussed in EPA Region 5 Biological Technical Assistance Group Bulletin No. 1 (EPA, 1996). Chemicals that exceed the ESVs, or for which no ESVs are available, will be retained as COPECs. The following ESVs, or ESV hierarchy (as noted), will be used for the ecological evaluation:

- ***Soil.*** Soil screening values will be selected using the following hierarchy: (1) EPA ecological soil screening levels (EPA, 2008); (2) *Preliminary Remediation Goals for Ecological Endpoints* (Efroymson et. al. 1997a); (3) EPA Region 5 ecological screening levels (EPA, 2003); (4) *Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Process* (Efroymson et al., 1997b); (5) *Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Terrestrial Plants* (Efroymson et al., 1997c); and (6) *Ecological Data Quality Levels* (EPA, 1999a). It should be noted that effects on heterotrophic processes may not be relevant to ecological receptors of concern at the sites.

- ***Groundwater.*** If groundwater is known to impact surface water at the sites, surface water ESVs will be used, as presented below.

- ***Surface Water.*** The lowest surface water screening value will be selected from the following three sources: (1) OEPA Water Quality Criteria (OAC Chapter 3745-1) for the protection of aquatic life; (2) *Preliminary Remediation Goals for Ecological Endpoints* (Efroymson et al., 1997a); and (3) *Ecological Data Quality Levels* (EPA, 1999a). A hierarchy will not be used because it would potentially eliminate important surface water COPECs, as OEPA water quality criteria do not consider food-chain effects.

- ***Sediment.*** Sediment screening values will be selected using the following hierarchy: (1) Consensus-based threshold effect concentration values (MacDonald, et al., 2000); (2) EPA Region 5 ecological screening levels (EPA, 2003); (3) *Ecological Data Quality Levels* (EPA, 1999a); (4) *Preliminary Remediation Goals for Ecological Endpoints* (Efroymson, et al., 1997a); and (5) *Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario* (Ontario Ministry of the Environment and Energy; 1993).

COPECs will be selected for further consideration in the SLERA if the MDC exceeds the available ESV. If no ESV is available, and the chemical is not eliminated as a COPEC due to other screening criteria, the constituent will be carried forward for further consideration in the SLERA.

2.6.7 Summary of COPEC Selection

A table of COPECs will be prepared for each medium with the following information:

- Chemical name
- Frequency of detection
- Range of detected concentrations
- Range of detection limits
- Arithmetic mean (average) of site concentrations
- Distribution type
- UCL of the mean of the concentration (only for chemicals selected as COPECs)
- Appropriate ESV
- The background screening concentrations
- COPEC selection conclusion: NO (with rationale for exclusion), or YES (selected).

Footnotes in the table(s) will provide the rationale for selecting or rejecting a chemical as a COPEC.

An evaluation of all of the constituents that were eliminated will be performed to determine whether any should be reinstated as COPECs due to other considerations. Examples of these exceptions include potential breakdown products, chemicals with detection limits greater than the ESV, chemicals known to have been used on site historically, and chemicals with high bioconcentration and/or bioaccumulation factors. Chemicals not eliminated using the screening procedures previously presented will be considered COPECs and will be quantitatively evaluated in the SLERA. The physical, chemical, and toxicological properties of the identified COPECs that are used directly in the exposure and effects analysis will be reviewed from the scientific literature and summarized in tables that present the pertinent information, with supporting citations. In addition, justification for the use of surrogate chemical data in the absence of direct chemical data for COPECs may be presented in the report tables and discussed in the text, as well.

2.7 Selection of Assessment Receptors

Assessment receptors will be selected for evaluation during this SLERA. In order to focus the exposure characterization portion of the SLERA on species or components that are the most likely to be affected and on those that, if affected, are most likely to produce greater effects in the on-site ecosystem, the selection process will focus on species, groups of species, or functional groups, rather than on higher organization levels such as communities or ecosystems.

Site biota will be organized into major functional groups. For terrestrial communities, the major groups are plants and wildlife, including terrestrial invertebrates, mammals, and birds. For aquatic and/or wetland communities, the major groups are flora and fauna, including vertebrates (water fowl and fish), aquatic invertebrates, and wetland/terrestrial mammals. Species presence and relative abundance will be determined during the site reconnaissance prior to identification of target species.

Primary criteria for selecting appropriate assessment receptors will include, but will not be limited to, the following:

- The assessment receptor will have a relatively high likelihood of contacting chemicals via direct or indirect exposure.
- The assessment receptor will exhibit marked sensitivity to chemicals.
- The assessment receptor will be a key component of ecosystem structure or function (e.g., importance in the food web, ecological relevance).
- The assessment receptor may be listed as rare, threatened, or endangered by a governmental organization; or the receptor will consist of critical habitat for rare, threatened, or endangered species.

Additional criteria for selection of assessment receptors will be used to identify species that offer the most favorable combination of characteristics for determining the implications of on-site contaminants. These criteria may include (1) limited home range, (2) role in local nonhuman food chains, (3) potential high abundance and wide distribution at the sites, (4) sufficient toxicological information available in the literature for comparative and interpretive purposes, (5) sensitivity to COPECs, (6) relatively high likelihood of occurrence onsite following remediation, (7) suitability for long-term monitoring, (8) importance to the stability of the ecological food chain or biotic community of concern, and (9) relatively high likelihood that they will be present at the sites or that habitats present at the sites could support the species.

It is important that sufficient toxicological information be available in the literature on the receptor species, or that a closely related species may be selected. While the ecological communities at the individual sites have species with many desirable characteristics for use as receptor species, not all of these species have been used extensively for toxicological testing.

Results of the assessment receptor selection process will be presented in detailed biological and

ecological descriptions called assessment receptor profiles (ARP). Additionally, the biologically relevant criteria used to select each assessment receptor will be discussed and summarized in the ARP. The ARPs will be included in the final SLERA report as an appendix.

2.8 Ecological Endpoint (Assessment and Measurement) Identification

The protection of ecological resources, such as habitats and species of plants and animals, is a principal motivation for conducting the SLERA. Potential ecological assessment and measurement endpoints will be proposed after the site reconnaissance and a thorough review of existing reports and site-related documents. The final assessment and measurement endpoints will be selected by agreement between risk assessors, risk managers, and regulatory agencies.

Unlike the human health risk assessment process, which focuses on individual receptors, the SLERA will focus on populations or groups of interbreeding nonhuman, nondomesticated receptors. In the SLERA process, the risks to individual receptors will be assessed only if they are protected under the Endangered Species Act, are species that are candidates for protection, or are species of special concern.

Given the diversity of the biological world and the multiple values placed on it by society, there is no universally applicable list of assessment endpoints. Suggested criteria that may be considered in selecting assessment endpoints suitable for a specific ecological risk assessment are (1) ecological relevance, (2) susceptibility to the contaminant(s), (3) accessibility to prediction and/or measurement, and (4) definable in clear, operational terms (Suter, 1993). Selected assessment endpoints will reflect environmental values that are protected by law, are critical resources, or have relevance to ecological functions that may be impaired. Both the entity and attribute will be identified for each assessment endpoint.

Assessment endpoints are inferred from effects to one or more measurement endpoints. The measurement endpoint is a measurable response to a stressor that is related to the valued attribute of the chosen assessment endpoint. It serves as a surrogate attribute of the ecological entity of interest (or of a closely related ecological entity) that can be used to draw a predictive conclusion about the potential for effects to the assessment endpoint.

Measurement endpoints for this SLERA will be based on toxicity values from the available literature and not statistical or arithmetic summaries of actual field or laboratory observations or measurements. When possible, receptors and endpoints will be concurrently selected by identi-

fying those that are known to be adversely affected by chemicals at the sites, based on published literature. COPECs for those receptors and endpoints will be identified by drawing on the scientific literature to obtain information regarding potential toxic effects of site chemicals to site species. This process will ensure that a conservative approach is taken in selecting endpoints and evaluating receptors that are likely to be adversely affected by the potentially most toxic chemicals at the sites. This information may be included in the ARP for appropriate receptors.

2.9 Ecological Site Conceptual Model

A pictorial representation of the exposure characterization will be prepared for the SLERA. This pictorial and any text necessary to clarify the representation will represent the ecological site conceptual model (ESCM). The ESCM will trace the contaminant pathways through both abiotic components and biotic food web components of the environment. The ESCM will present all potential exposure pathways and will identify those pathways that are complete and incomplete. The ESCM will clearly identify the relationship between the measurement and assessment endpoints. It will be used as a tool for judging the appropriateness and usefulness of the selected measurement endpoints in evaluating the assessment endpoints, and for identifying sources of uncertainty in the exposure characterization. All existing data will be qualitatively reviewed for quality, usefulness, and uncertainty.

3.0 Exposure Characterization

An estimate of the nature, extent, and magnitude of potential exposure of assessment receptors to COPECs that are present at or migrating from the sites, considering both current and reasonably plausible future uses of the sites will be presented in the SLERA Exposure Characterization. Exposure and chemical uptake will be modeled to produce upper-bound exposure estimates. Exposure characterization is critical in further evaluating the risks of compounds identified as COPECs during the screening process (Section 2.6). The exposure assessments will be conducted by characterizing the magnitude (concentration) and distribution (locations) of the contaminants detected in the media sampled during the investigation, evaluating pathways by which chemicals may be transported through the environment, and determining the points at which organisms found in the study area may contact contaminants.

3.1 Exposure Analysis

An exposure analysis will be performed in the SLERA, which will combine the spatial and temporal distribution of the ecological receptors with those of the COPECs to evaluate exposure. The exposure analysis will focus on the chemical amounts that are bioavailable, and the means by which the ecological receptors are exposed (e.g., exposure pathways). The focus of the analysis will be dependent on the assessment receptors being evaluated as well as the assessment and measurement endpoints.

Exposure pathways consist of four primary components: source and mechanism of contaminant release, transport medium, potential receptors, and exposure route. A chemical may also be transferred between several intermediate media before reaching the potential receptor. All of these components will be addressed within the SLERA. If any of these components is not complete, then contaminants in those media do not constitute an environmental risk at that specific site. The major fate and transport properties associated with typical site contaminants will be outlined. These properties directly affect a contaminant's behavior in each of the exposure pathway components.

Calculation of plant exposure is not necessary, as the plant toxicity data are expressed in concentration in the growth medium. For terrestrial faunal receptors, calculation of exposure rates relies upon determination of an organism's exposure to COPECs found in surface water, surface soil, and sediment. Exposure rates for terrestrial wildlife receptors will be based solely

upon ingestion of contaminants from these media and consumption of other organisms. Given the scarcity of available data for wildlife dermal and/or inhalation exposure pathways, potential risk from these pathways will not be estimated. In addition, these pathways are generally considered to be incidental for most species, with the possible exceptions of burrowing animals and dust-bathing birds.

Daily doses of COPECs for vertebrate receptors will be calculated using standard exposure algorithms. These algorithms incorporate species-specific natural history parameters (i.e., feeding rates, water ingestion rates, dietary composition, etc.) and also use site-specific area use factors, as follows:

$$Total\ Daily\ Dose = \left(\frac{\left(\left[Soil_j * IR_{soil} \right] + \left[Water_j * IR_{water} \right] + \left[\sum_{i=1}^{N} B_{ji} * P_i * IR_{food} \right] \right)}{Body\ Weight} \right) * AUF$$

where:

$Soil_j$	=	Concentration of COPEC "j" in soil
$Water_j$	=	Concentration of COPEC "j" in surface water
B_{ji}	=	Concentration of COPEC "j" in food type "i"
IR_{soil}	=	Soil ingestion rate
IR_{water}	=	Surface water ingestion rate
IR_{food}	=	Food ingestion rate
P_i	=	Proportion of food $type_i$ in receptor diet
AUF	=	Area use factor (equal to area of exposure unit/home range of receptor
Body Weight	=	Body weight of receptor.

Sediment may replace soil in the above equation for aquatic or semiaquatic receptors.

The first step in estimating exposure rates for terrestrial wildlife involves the calculation of feeding and drinking rates for site receptors. EPA (1993) includes a variety of exposure information for a number of avian, herptile, and mammalian species. Information regarding feeding and drinking rates and dietary composition are available for many species, or may be estimated using allometric equations (Nagy, 1987). Data will be gathered on incidental ingestion of soil and will be incorporated for the receptor species. This information will be summarized and documented in the ARPs. Literature values for animal-specific sediment ingestion will be

used, if available. However, such values generally are not available in the literature. Where sediment ingestion rates cannot be found, the animal-specific incidental soil ingestion rate will be used for sediment ingestion as well, if the receptor's life history profile suggests a significant aquatic component (e.g., raccoons' use of surface water in foraging activities).

Algorithms will be evaluated for calculating exposure for terrestrial vertebrates that account for exposure via ingestion of contaminated water, incidental ingestion of contaminated soil, and ingestion of plants grown in contaminated soil. Singular algorithms will be developed for soil-to-plant uptake and for animal bioaccumulation. An assessment exposure via uptake by carnivores will also be included.

Ecological routes of exposure for biota may be direct (bioconcentration) or through the food web via the consumption of contaminated organisms (biomagnification). Direct exposure routes include dermal contact, absorption, inhalation, and ingestion. Examples of direct exposure include animals incidentally ingesting contaminated soil or sediment (e.g., during burrowing or dust-bathing activities), animals ingesting surface water, plants absorbing contaminants by uptake from contaminated sediment or soil. and the dermal contact of aquatic organisms with contaminated surface water or sediment.

Food web exposure can occur when terrestrial or aquatic fauna consume contaminated biota. Examples of food web exposure include animals at higher trophic levels consuming plants or animals that bioaccumulate contaminants. The concepts of bioconcentration, bioaccumulation, and biomagnification are used throughout this document. These terms are defined by EPA (1997) as follows:

- ***Bioaccumulation.*** General term describing a process by which chemicals are taken up by an organism either directly from exposure to a contaminated medium or by consumption of food containing the chemical.

- ***Bioconcentration.*** A process by which there is a net accumulation of a chemical directly from an exposure medium into an organism.

- ***Biomagnification.*** Result of the process of bioaccumulation and biotransfer by which tissue concentrations of chemicals in organisms at one trophic level exceed tissue concentrations in organisms at the next lower trophic level in a food chain.

Contamination of biota could result from exposure to one or more COPECs. Bioavailability is

an important contaminant characteristic that influences the degree of chemical-receptor interaction. Bioavailable compounds are those that a receptor can take in from the environment. Bioavailability of a chemical is a function of several physical and chemical factors.

Per EPA (1995) guidance, aquatic bioaccumulation factors (BAF) will be estimated by one of four methods (in order of preference):

- A measured BAF for an inorganic or organic chemical derived from a field study
- A predicted BAF for an organic chemical derived from a field-measured biota-sediment accumulation factor
- A predicted BAF for an inorganic or organic chemical derived from a laboratory-measured bioconcentration factor and a food chain multiplier (FCM)
- A predicted BAF for an organic chemical derived from a log octanol-water partition (K_{ow}**)** and an FCM.

The EPA guidance notes, however, that for chemicals for which no K_{ow} is available and for which no bioconcentration factor is calculable, a default FCM of 1.0 should be used. Thus, for inorganic chemicals not thought to biomagnify and/or for which no literature value is available, this value of 1.0 will be used at each trophic level.

For the current SLERA, site-specific BAFs, from the data reflected as per the Red Water Ponds Phase II ecological risk assessment work plan (IT, 2000) will be given the highest preference, when appropriate, followed by BAFs recommended in EPA (2008). To supplement these values, BAFs will be estimated for organic constituents using other sources or calculations identified in the literature, such as the log K_{ow} relationships developed by Travis and Arms (1988). Particular emphasis will be given to BAF sources recommended in OEPA (2008) guidance.

Media-Specific Exposure Pathways. Exposure to four categories of environmental media will be addressed in the SLERA, as discussed in the following subsections.

Soil Exposure Pathway. Soil exposure pathways are potentially important for terrestrial plants and animals at the sites. For non-burrowing animal exposure, soil samples obtained from a depth of 0 to 1 foot bgs will be considered, as this would be the point of exposure. For

burrowing animals, soil samples obtained from a depth of 0 to 6 feet bgs will be considered. For plant exposure, soil samples taken from 0 to 6 feet bgs (or the water table surface) will be considered, because most feeder roots are located within this depth.

Environmental conditions such as soil moisture, soil pH, and cation exchange capacities significantly influence whether potential soil contaminants remain chemically bound in the soil matrix or can be chemically mobilized (in a bioavailable form) and released for plant absorption. Generally, neutral to alkaline soils (soil pH of 6.5 or greater) restrict the absorption of toxic metals, making pathway completion to plants difficult. Literature values for soil-to-plant transfer rates for inorganic and organic soil contaminants and for organic soil contaminants will be used unless contaminant-specific information is available.

Sediment Exposure Pathway. Sediment generally consists of soil or other material settled out of suspension in surface water or native soils underlying flowing or standing surface water bodies. Potential contaminant sources for sediment include buried or stored waste, and contaminated surface water, groundwater, and soil. The release mechanisms include surface water runoff, groundwater discharge, and airborne deposition. Potential receptors of chemicals in contaminated sediment include aquatic flora and fauna. Direct exposure routes for contaminated sediment include contact by benthic-dwelling organisms such as catfish, uptake by aquatic flora, and ingestion by aquatic fauna. Indirect exposure pathways from sediment include consumption of bioaccumulated contaminants by consumers in the food chain. Chemical bioavailability of many nonpolar organic compounds (e.g., polychlorinated biphenyls and pesticides) decreases with increasing concentrations of total organic carbon in the sediment; however, these compounds can still bioaccumulate up the food chain (Landrum and Robbins, 1990).

Surface Water Exposure Pathway. Surface water represents a potential transport medium for COPECs. Potential sources for contaminated surface water include buried or stored waste, stored or spilled fuel, contaminated soil and groundwater, and deposition of airborne contaminants. The release mechanisms include surface runoff, leaching, and groundwater seepage. Potential receptors of contaminated surface water include terrestrial and aquatic fauna and aquatic flora. Exposure routes for contaminated surface water include ingestion by terrestrial fauna and uptake and absorption by aquatic flora and fauna. Consumption of bioaccumulated contaminants constitutes a potential indirect exposure pathway for faunal receptors. Chemical

bioavailability of some metals and other chemicals is controlled by water hardness, pH, and total suspended solids.

Groundwater Exposure Pathway. Groundwater represents a potential transport medium for COPECs. Potential contaminant sources for groundwater include contaminated soil and buried or stored waste. The release mechanism for contaminants into groundwater is direct transfer of contaminants from waste materials to water as water passes through the materials.

Groundwater itself is not an exposure point, although contaminant transport along the shallow groundwater pathway may be considered an exposure route to aquatic life, wetlands, and some wildlife where the groundwater discharges to surface water. This pathway is of importance to aquatic and wetland receptors if groundwater is found to be discharging to surface water. It should be noted that groundwater will not be evaluated as a medium of concern in the SLERA when surface water data are available or when groundwater is not discharging to the surface.

3.2 Exposure Characterization Summary

At the conclusion of the exposure characterization, the estimated chemical intakes for each exposed receptor group under each exposure pathway and scenario will be presented in tabular form. The presentation will include an identification of all pertinent factors. These intake estimates will be combined with the COPEC toxicity values (discussed in the following chapter) to derive estimates and characterize potential ecological risk. The uncertainties associated with the estimation of chemical intake will be summarized at the conclusion of the exposure characterization. The basis for each uncertainty will be identified, with the degree of uncertainty estimated qualitatively (low, medium, or high) or quantitatively, and the impact of the uncertainty will be estimated qualitatively (overestimate or underestimate, as appropriate).

4.0 Ecological Effects Characterization

The ecological effects characterization will include the selection of literature benchmark values and the development of toxicity reference values (TRV).

4.1 Selection of Literature Benchmark Values

Appropriate sources for literature benchmark values will be consulted for toxicological information, such as (1) *Toxicological Benchmarks for Wildlife* (Sample, et al., 1996), *Development of Toxicity Reference Values for Conducting Ecological Risk Assessments at Naval Facilities in California* (Engineering Field Activity, West, 1998), *Review of the Navy - EPA Region 9 BTAG Toxicity Reference Values for Wildlife* (CH2M Hill, 2000); and (2) lethal dose 50 percent values from data bases such as the Registry of Toxic Effects Concentrations (extrapolated to chronic no-observed-adverse-effect level [NOAEL] or lowest-observed-adverse-effect level [LOAEL] values using recommended Tri-Service [Wentsel, et al., 1996] and OEPA [2008] uncertainty factors). The level of effort will be limited to documents that summarize the available ecotoxicological information and will not include review of the primary toxicological literature (i.e., details of toxicity test conditions to determine validity of the tests performed will not be reviewed).

The equilibrium partitioning approach has been used by EPA and Ontario Ministry of the Environment and Energy in the preparation of sediment quality criteria for the protection of aquatic life. These criteria will be used, where available, to assess sediment risks to aquatic receptors.

4.2 Development of Reference Toxicity Values

TRVs for the site will be developed or determined. These TRVs will focus on the growth, survival, and reproduction of species and/or populations. Empirical data may be available for the specific receptor-endpoint combinations in some instances. However, for some COPECs, data on surrogate species and/or on endpoints other than the NOAEL and LOAEL may have to be used. The NOAEL is a dose of each COPEC that will produce no known adverse effects in the test species. The NOAEL was judged to be an appropriate toxicological endpoint since it would provide the greatest degree of protection to the receptor species. The LOAEL will be used as a point of comparison for decision-making for risk management purposes. In addition, in instances where data are unavailable for a site-associated COPEC, toxicological information for

surrogate chemicals may be used. Safety factors will be used to adjust for these differences and extrapolate effects to the site's receptors at the NOAEL and/or LOAEL endpoint, as described in the following paragraphs.

Toxicity information pertinent to identified receptors will be gathered for those analytes identified as COPECs. Because the measurement endpoint will range from the NOAEL to the LOAEL, preference will be given to chronic studies noting concentrations at which no adverse effects were observed and ones for which the lowest concentrations associated with adverse effects were observed. As previously noted, where data are unavailable for the exposure of a receptor to a COPEC, data for a surrogate chemical (e.g., endrin for endrin aldehyde) will be gathered for use in the SLERA.

Using the relevant toxicity information, TRVs will be calculated for each of the COPECs. TRVs represent NOAELs and LOAELs with safety factors incorporated for toxicity information derived from studies other than no-effects or lowest-effects studies, and studies on species other than the receptors selected for this risk assessment (Figure 4-1). TRVs will be obtained from the open literature including the wildlife toxicity assessments and terrestrial toxicity database from the U.S. Army Center for Health Promotion and Protective Medicine available at (http://chppm-www.apgea.army.mil/tox/HERP.aspx) and EPA's *Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (EPA, 1999b). Because NOAELs for the selected wildlife receptor species will most likely be based on NOAELs from test species, the latter will be converted to NOAELs specific to the selected wildlife receptors using a power function of the ratio of body weights, as described by Sample, et al. (1996). A body weight scaling factor of 0.25 will be used for mammals, whereas a body weight scaling factor of 0 will be used for birds, making the $NOAEL_W$ for birds the same as the $NOAEL_T$, as shown below (OEPA, 2008):

$$NOAEL_W = NOAEL_T \left(\frac{BW_T}{BW_W} \right)^s$$

where:

$NOAEL_w$	=	the No Observed Adverse Effect Level for the wildlife indicator species (mg/kg-day)
$NOAEL_T$	=	the No Observed Adverse Effect Level for the test species (mg/kg-day)
BW_T	=	the body weight of the test species (kg)
BW_W	=	the body weight of the wildlife indicator species (kg)

s	=	a body weight scaling factor (s = 1/4 for mammals and s = 0 for birds).

Exposure rate TRVs provide a reference point for the comparison of toxicological effects upon exposure to a contaminant. To complete this comparison, receptor exposure to site contaminants must be calculated, or as in the case of plant receptors, exposure is simply calculated as the soil concentration.

5.0 Risk Characterization

The risk characterization phase integrates information on exposure, exposure-effects relationships, and defined or presumed target populations. The result is a determination of the likelihood, severity, and characteristics of adverse effects of environmental stressors present at a site. A semi-quantitative approach will be taken to estimating the likelihood of adverse effects occurring as a result of exposure of the selected site receptors to COPECs. For plant and invertebrate receptors, risk characterization will essentially be the comparison to the benchmark values presented in Section 2.6.6. For vertebrate receptors, TRVs and exposure rates will be calculated and used to generate hazard quotients (HQ) (Wentsel, et al., 1996) by dividing the receptor exposure rate for each contaminant by the calculated TRV. HQs are a means of estimating the potential for adverse effects to organisms of a contaminated site and for assessing the potential that toxicological effects will occur among site receptors.

5.1 Risk Estimation

The potential hazard associated with the sites will be estimated. The hazard estimation will be performed through a series of quantitative HQ calculations that compare receptor-specific exposure values with TRVs. The simple HQ ratios may be summed, where appropriate and scientifically defensible, to provide hazard index estimates for all chemicals and exposure pathways for a given receptor (e.g., polynuclear aromatic hydrocarbons, phthalates, structurally similar organochlorine pesticides). For a given receptor, only HQs for those chemicals that have a similar mode of toxicological action will be summed.

Because of the conservative assumptions used to determine receptors' daily doses of COPECs and in generating TRVs, HQs greater than 1.0 are not unexpected, and should not be interpreted to mean that an adverse effect is occurring, or has occurred in the past. Therefore, the HQs will be evaluated using guidelines for assessing the potential risk posed from contaminants. HQs less than or equal to 1.0 present no probable risk; HQs greater than 1.0 but less than 10 present a low potential for environmental effects; HQs from 10 up to, but less than 100, present a significant potential that effects could result from greater exposure; and HQs greater than 100 present the highest potential for expected effects (Wentsel, et al., 1996). Although OEPA considers HQs greater than 1.0 to be potentially significant, it should be noted that HQs are not measures of risk, population-based statistics, or linearly scaled statistics. Thus, an HQ above 1, even exceedingly so, does not necessarily mean that there is even one individual expressing the

toxicological effect associated with a given chemical to which it was exposed (Tannenbaum, 2003; Bartell, 1996).

5.2 Uncertainty Analysis

The results of the SLERA will be influenced to some degree by variability and uncertainty. In theory, investigators might reduce variability by increasing sample size of the medium or species sampled. Alternatively, uncertainty within the risk analysis can be reduced by using species-specific and site-specific data (i.e., to better quantify contamination of media, vegetation, and prey through direct field measurements, toxicity testing of site-specific media, field studies using site-specific receptor species). Detailed media, prey, and receptor field studies are costly; thus, the preliminary scoping and predictive analyses of risk are conducted to limit the potential use of these resource-intensive techniques to only those COPECs that continue to show a relatively high potential for ecological risk. Because assessment criteria were developed based on conservative assumptions, the results of the screening and predictive assessments will err on the side of conservatism. This has the effect of maximizing the likelihood of accepting a false positive (Type I error: the rejection of a true null hypothesis) and simultaneously minimizing the likelihood of accepting a true negative (Type II error: the acceptance of a false null hypothesis). The use of soil data from 0 to 6 feet bgs may overestimate ecological effects, because many ecological receptors are only exposed to shallower soils. The uncertainty analysis will thus assess the soil depth of elevated concentrations of COPECs identified as risk drivers, and will evaluate the significance of these findings on the results of the SLERA (e.g., if COPEC hot spots only occur at deeper soil depths, realistic ecological exposure could be expected to be minimal).

A number of factors contribute to the overall variability and uncertainty inherent in ecological risk assessments. Variability is due primarily to measurement error; laboratory media analyses and receptor study design are the major sources of this kind of error. Uncertainty, on the other hand, is associated primarily with deficiency or irrelevancy of effects, exposure, or habitat data to actual ecological conditions at the sites. Species physiology, feeding patterns, and nesting behavior are poorly predictable; therefore, all toxicity information derived from toxicity testing, field studies, or observation will have uncertainties associated with them. Laboratory studies conducted to obtain site-specific, measured information often suffer from poor relevance to the actual exposure and uptake conditions onsite (i.e., bioavailability, exposure, assimilation, etc., are generally greater under laboratory conditions as compared to field conditions). Calculating an estimated value based on a large number of assumptions is often the only alternative to the accurate (but costly) method of direct field or laboratory observation, measurement, or testing.

Finally, habitat- or site-specific species may be misidentified if, for example, the observational assessment results are based on only one brief site reconnaissance performed on a relatively large site.

The calculation of HQs also introduces uncertainty. The following limitations associated with HQs are noted and will be briefly addressed in the final SLERA report:

- HQs are not measures of risk.
- HQs are not population based.
- HQs are not linearly scaled.
- HQs are often produced that are unrealistically high and toxicologically impossible.
- Trace soil concentrations of inorganic chemicals (including concentrations well below background levels) can lead to HQ threshold exceedances.

The uncertainty analysis will include a discussion of the assumptions made for the SLERA, including the direction of bias caused by each assumption (i.e., if the uncertainty results in an overestimate or underestimate of risk), the likely magnitude of impact and, if possible, a description of recommendations for minimizing the identified uncertainties if the SLERA progresses to higher level assessment phases (EPA, 1997).

5.3 Risk Description

As part of the risk description, the following will be completed: (1) summarize the ecological risk associated with the sites; and (2) interpret the ecological significance, which describes the magnitude of the identified risks and the accompanying uncertainty. The effect of additional data or analyses on uncertainty will also be discussed. A weight-of-evidence approach will be used to interpret the ecological significance of the findings.

6.0 Risk Summary and Conclusions

The potential adverse ecological effects associated with releases from the site will be summarized. This summary will be supported by the steps performed as described in the previous sections. Additionally, recommendations for further investigations will be made if appropriate. The information presented in the SLERA, including calculated HQs and their associated uncertainties, will be used in the feasibility study to develop cost-effective site-specific remedial action objectives, if remedial action for the protection of ecological receptors is warranted.

7.0 References

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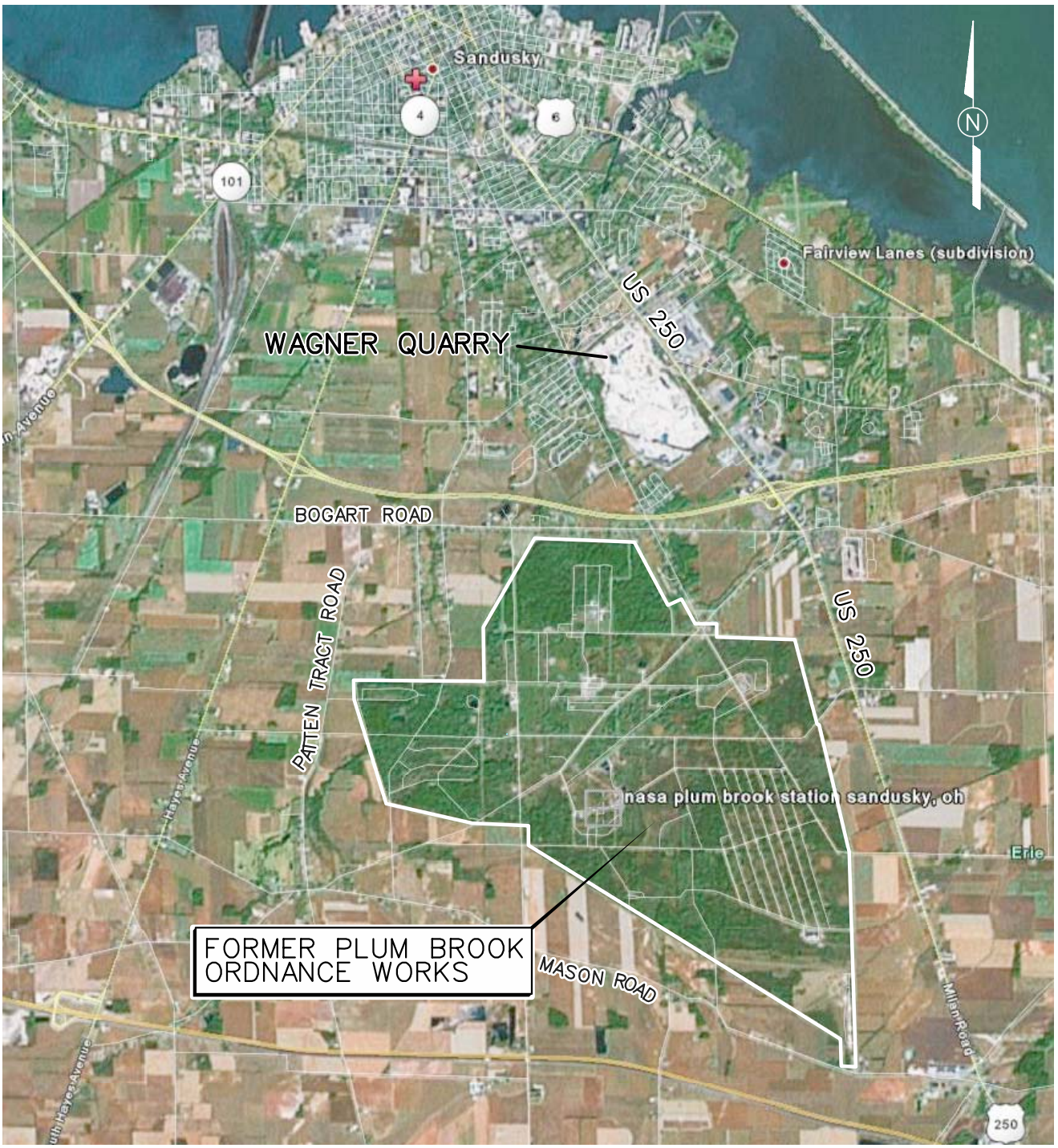
FIGURES



NOT TO SCALE

FIGURE 1-1
PBOW VICINITY MAP

WWTP1 &3 AND AP1 &3 BASELINE HUMAN HEALTH RISK ASSESSMENT WORK PLAN
FORMER PLUM BROOK ORDNANCE WORKS
NASA PLUM BROOK STATION
SANDUSKY, OHIO

Shaw® Shaw Environmental & Infrastructure, Inc.

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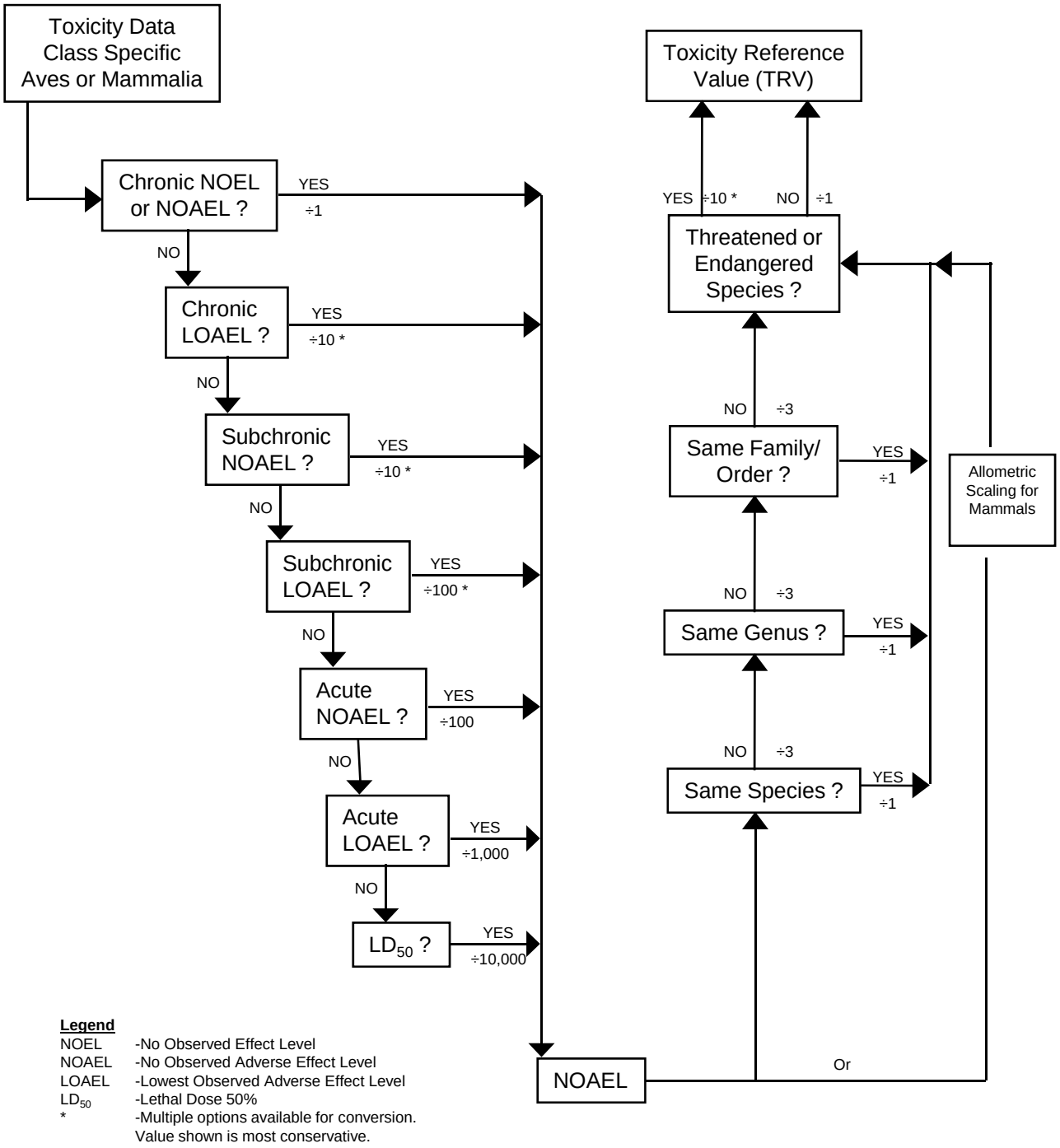
Figure 4-1

Procedural Flow Chart for Deriving Toxicity Reference Values (TRV) from Class-Specific Toxicity Data Plum Brook Ordnance Works, Sandusky, Ohio



Legend

NOEL	-No Observed Effect Level
NOAEL	-No Observed Adverse Effect Level
LOAEL	-Lowest Observed Adverse Effect Level
LD_{50}	-Lethal Dose 50%
*	-Multiple options available for conversion. Value shown is most conservative.

Credit: Adapted from Ohio EPA (OEPA), 2008, ***Guidance for Conducting Ecological Risk Assessments***, DERR-00-RR-031, April.

Responses to Comments
Work Plans, Baseline Human Health Risk Assessment (BHHRA) and Screening Level Ecological Risk Assessment (SLERA) for Waste Water Treatment Plants 1 and 3 and Ash Pits 1 and 3, Plum Brook Ordnance Works, Sandusky, Ohio, Dated June 15, 2009

Comments by Janusz Byczkowski, Risk Assessor, Ohio Environmental Protection Agency, received July 6, 2009.

Comment 1: **BHHRA Section 4.1, Page 4-3, Equation 4.1. These Documents should be revised to correct some inaccuracies and/or typing/editing errors.**

For example, the BHHRA Document states:
"…Eq. 4.1
(P(d) – P(0))/(1 – P(0))…"

By definition, an equation is a mathematical statement, in symbols, that two things are exactly the same (or equivalent), and thus it should contain at least two expressions connected by an equal sign (=), In the Eq. 4.1, the left site expression and the equality sign are missing. Please correct equation 4.1.

Response 1: "SF =" will be added to the left side of the equation and defined as "slope factor" in the legend. The BHHRA will be re-checked regarding unspecified typing/editing errors.

Comment 2: **BHHRA, Section 4.1. Page 4-1, Line 15. This Document states: "…EPA (1986) recognizes six weight-of-evidence group classifications for carcinogenicity…"**

The EPA (1986) reference is obsolete. According to U.S. EPA Integrated Risk information System (IRIS program), currently used final cancer guidelines have been published by EPA in March 2005:

"The U.S. EPA (2005) guidelines revise and replace the U.S. Environmental Protection Agency's (EPA's, or the Agency's) Guidelines for Carcinogen Risk Assessment, published in 51 FR 33992, September 24, 1986 and the 1999 interim final guidelines".

Please use standard carcinogenicity hazard descriptors, as defined in the current U.S. EPA (2005) final guidelines: "There are five recommended standard hazard descriptors: "Carcinogenic to Humans," "Likely to Be Carcinogenic to Humans," "Suggestive Evidence of Carcinogenic Potential," "Inadequate Information to

Assess Carcinogenic Potential," and "Not Likely to Be Carcinogenic to Humans".

Reference:
U.S. EPA, 2005, Guidelines for Carcinogen Risk Assessment. EPA/630/P-03/001B March 2005, Risk Assessment Forum U.S. Environmental Protection Agency Washington, DC. On line: http://nepis.epa.gov/Adobe/PDF/P100079H.PDF.

Response 2: The referenced 2005 guidance will be used in the text. Mention of letter-based weight-of-evidence will also be provided, as many of the Integrated Risk Information System (IRIS) records still use these descriptors. Also, for easy of review, brevity, and consistency with previous PBOW document, the letter-based descriptors will be used in the report tables with footnotes provided to define the equivalence of these values based on the current EPA 2005 system.

Comment 3: **BHHRA, Figure 2-1, and Section 2.4.3.1. Page 2-6, Line 25. The paradigm presented in Figure 2-1 does not follow the OEPA – DERR RI/FS Program, and the decision tree, as presented, could be misleading. Consequently, in the text, the background concentration has been described as "…95th percent upper tolerance limit of the background data set…" Such a background calculation methodology is not recommended by OEPA-DERR.**

For example, if "population testing (e.g., WRS)" shows that the concentrations of analyte in area of concern are always below background results, then the sample of data from area of concern and the sample of data from background area belong to two different populations. Such a result of "population testing" means (paradoxically) that the analyte is NOT background-related, and thus it should NOT be screened from risk evaluation. If the same distribution patterns are demonstrated for several analytes, then the area of concern is not related to background area, or the background site is inadequate for use as a reference area. This paradox has been avoided in the background methodology developed by OEPA-DERR (2004a and 2004b).

It is recommended that the Figure 2-1 should not be included at or followed in preparing the Risk Assessment report based on this Work Plan. Instead, the OEPA – DERR (2004c) should be applied along with other relevant technical decision documents (TDCs).

References:
OEPA – DERR, 2004a, Methodology for Evaluating Site-specific Background Concentrations of Chemicals, Technical Decision Compendium, 14 April 2004. On-line: http://www.epa.state.oh.us/derr/rules/Methodology.pdf
OEPA-DERR, 2004b, Background Calculation Methodology, Guidance DERR-00-RR-039P, 30 June 2004, Final. On-line: http://www.epa.state.oh.us/derr/rules/RR-039_public.pdf
OEPA-DERR, 2004c, Use of Risk-Based Numbers in the Remedial Response Process Overview (revised June 28, 2005). Available on-line: http://www.epa.state.oh.us/derr/rules/RR-038.pdf.

Response 3: The soil background data set and evaluation approach for all PBOW sites is described in the TNT Areas A and C BHHRA (IT, 2001b). This approach was developed as a PBOW Team effort, with input from USACE and OEPA risk assessors Laurie Moore and Bonnie Buthker. After evaluating numerous background statistics for soil (e.g., mean, median, UCLs and UTLs), the OEPA risk assessors and the rest of the PBOW Team deemed the 95 percent UTL or maximum detected, whichever is less, to be the appropriate background statistic to use during background screening of metals at PBOW areas of concern. The consensus to use this approach for soil is documented in the May 2000 PBOW Team Meeting minutes.

The development of the background evaluation approach for groundwater was likewise a team effort developed by the USACE and OEPA, including the risk assessors. The approach and the background groundwater data set are described in Appendix M of the 2004 Groundwater Data Summary and Evaluation Report (Shaw, 2005). If the population testing or other background evaluation results show that the site concentrations are found to be less than those of the background data set, this would be discussed in the uncertainty analysis."

Figure 2-1 was included in response to internal review prior to approval of a previous site work plan. This figure is consistent with the data screening protocol agreed upon during PBOW Team meetings and considered as a consensus agreement among the PBOW Team, including USACE, OEPA, and NASA. It is also consistent with the OEPA-DERR (2004c) document cited by this comment. As requested, this figure will be deleted from the work plan.

Comment 4: **BHHRA, Table 3-2, Page 1. This document states: "Exposure duration (ED), years… […] On-Site Resident […] Adult: 24a…" […] "Averaging time, noncancer (AT), daysb […] Adult: 8760…"**

and the footnote:
"bCalculated as the product of ED (years) x 365 days/year"

Please explain in this document that the total exposure duration (ED) for adult in calculations of health hazard is 30 years, and for calculating cancer risk, the ED is 24 + 6 = 30 years too. Please adjust AT appropriately and provide necessary explanations.

According to quoted in the footnote "b" EPA (1989) RAGS A, the default noncarcinogenic exposure duration for adult resident should be 30 years. Only for carcinogens, it should be taken into account, in the calculations of lifetime excess cancer risk for adult, that the first 6 years of the lifetime exposure was encountered as a child. Therefore, applying the product from the footnote "b" to the noncancer AT would yield 30 x 365= 10950 days rather than 8760.

Response 4: Consistent with EPA (1991) guidance, the residential exposure scenario is broken into two life stages: young child (6-year duration, ages 1 through 6) and adult (24 years). This has been done for all approved PBOW risk human health risk assessment work plans and reports dating back to 1998. To clarify the current work plan, the sentence extending from page 3-13 to 3-14 in the draft will be revised consistent with the following: "The on-site residential scenario is evaluated *assuming a 30-residential exposure scenario*, considering exposure to *a resident as a* young child (*6-year duration*, ages 1 through 6 years) *through adult portion of life spent at this residence (24-year duration) (EPA, 1991a)."* The ensuing text will also be revised to state noncancer hazards for adult residents *will* be quantitatively evaluated, consistent with the most recent PBOW human health risk assessment work plans and reports.

As depicted by EPA (1989) in Exhibit 6-14 of RAGS A, the recommended ED of 30 years describes the 90th percentile of time spent at one residence. This is consistent with the overall 30-year residential exposure scenario described in the draft work plan. If only the adult life stage were being considered, then the averaging time would be 10,950 days as asserted by the reviewer. However, Exhibit 6-14 of RAGS A states that the AT is based on the pathway-specific period of exposure or ED × 365 days/year. Thus, for the child, the ED of 6 years results in an AT of 2,190 days (6 × 365) and an ED of 24 years for the adult results in an AT of 8,760 days. (24 × 365). Although it might be said that the overall ED for the resident is 30 years, separate AT values must be used for the noncancer evaluation of the child and adult or a nonconservative (i.e., low) bias will be introduced into the noncancer hazard estimates.

It does not appear that RAGS A clearly states, as the reviewer suggests, that a 30-year ED is to be used specifically for the adult life stage of the resident. Even if that were the case, the Human Health Evaluation Manual Supplemental Guidance (EPA, 1991) clearly states, "…where exposure factors differ, values presented in this guidance supersede those presented in HHEM Part A (i.e., "RAGS A")." As stated above, EPA (1991) clearly describes that the 30-year residential exposure period is to be divided into two parts, a 6-year duration for young children and a 24-year duration for older children and adults.

Additionally, it is noted that because the ED appears in the numerator and the AT (ED × 365 for noncarcingens) appears in the denominator for each of the basic intake equation. Because the AT is simple the ED multiplied by 365 for the evaluation of noncancer hazard, it makes absolutely no mathematical difference as to what AT value is used. Whatever the ED value, the ED÷AT = 1/(365 days/year) and has absolutely no effect on the intake estimate.

Comment 5: BHHRA, Table 3-2, Page 3. This Document states:
"…Body surface area exposed to sediment (SAsd) cm2…" […]
"Child: 5,700g […] Adult 2,800g…"
and then:
"…Body surface area exposed to surface water (SAsw) cm2…" […]
"Child: 2,800h […] Adult 7,000c…"

A reality check should be performed on exposure parameters compiled from different sources. For example, even though according to RAGS E, for dermal contact with sediment the available body surface area for child includes the same body parts as for adults plus feet, under no circumstances this could result in the surface area of child twice greater than that the adult. Moreover, according to 15 kg body weight, the child age could be only about 3 years, rather than 6 years old.

In order to realistically simulate any exposure scenario in children, it is important to match all the parameters either with the body weight or with the specific, narrow age group. In children, physiological parameters such as body weight, surface area (e.g., see: Burmaster, 1989), inhalation rate, water intake, etc., are highly correlated with each other and dependent on age. To avoid introducing additional variability to the exposure model, and consequently increasing the uncertainty in the estimated risk, all numerical exposure calculations should either use parameters for narrow age groups or age-dependent growth functions.

Mixing physiological parameters for different age groups in the resident child and hunter's child scenarios do not appear to be scientifically sound. The exposure parameters should be changed to match the selected group of children, in accordance with the "Child-Specific Exposure Factors Handbook" (U.S. EPA, 2002). Please revise numerical values and correct the exposure parameters, according to consistent scenarios for specified receptors.

Reference:

Burmaster, D. E, 1998, "Lognormal distributions for skin area as a function of body weight," Risk Analysis, 18(1):27-32.
U.S. EPA, 2002, Child-Specific Exposure Factors Handbook (Interim Report 2002). U.S. Environmental Protection Agency, Office of Research and Development, National Center for Environmental Assessment, Washington Office, Washington, DC, EPA-600-P-00-002B, 2002. Available on-line: http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=55145.

Response 5: The exposed body surface areas for the resident child and adult were inadvertently reversed in the Table 3-2. These will be corrected in the final table. Values of 5,700 square centimeters (cm^2) for the adult and 2,800 cm^2 for the child are portrayed correctly on page 3-15 of the draft text, with appropriate explanation and reference to guidance documents. Note that as stated, these are the default reasonable maximum exposure surface area values shown in the RAGS E (EPA, 2004) guidance.

The use of a 15 kg body weight follows current EPA guidance (e.g., EPA, 2004a) and this approach has been used in EPA guidance since the early 1990s (e.g., EPA, 1991). It is also part of the paradigms used in screening values that have been adopted by the OEPA-DERR, including the Regional Screening Values Oak Ridge National Laboratory-EPA (2009) and EPA Region 9 Preliminary Screening Values (EPA, 2004b). Further, the use of the 15 kg value for the young child is consistent with the evaluation of other PBOW sites and is documented in the TNT Areas A and C human health risk assessment work plan (IT, 2001a), which has been regarded as the agreed consensus protocol for risk assessment work by the PBOW Project Team.

Note that the use of a 15 kg body weight represents the approximate midpoint in the range between ages 1 through 6. The mean body weights for these age groups are 11 kg (age 1) through 23 kg (age 6) (EPA, 1997). The arithmetic midpoint of this range is 17 kg and that the logarithmic midpoint is 16 kg. Because a lower weight results in a higher estimate of risk, the value of 15 kg appears appropriately conservative for this age range.

Comment 6: **SLERA, Section 2.6.4, Page 2-7. This Document states: "…background soil samples […] 95th percent upper tolerance limit […] if the background data and the site data are drawn from the same population…"**

OEPA-DERR does not recommend the usage of 95th percent UTL for determination of background levels. For comments, see above: BHHRA # 3. Please follow the background methodology as described in OEPA-DERR (2004a, b).

Response 6: The background approach was agreed upon by the PBOW Project Team prior to the referenced OEPA documents. The agreed upon approach is described in the TNT Areas A and C BHHRA (IT, 2001b) and Appendix M of the 2004 Groundwater Data Summary and Evaluation Report (Shaw, 2005). Please also refer to the response to Comment No. 3.

List of References Used in Responses

IT Corporation (IT), 2001a, ***Baseline Human Health Risk Assessment and Ecological Risk Assessment Work Plan***, TNT Areas A and C, Final, Plum Brook Ordnance Works, Sandusky, Ohio, May.

IT Corporation (IT), 2001b, ***Baseline Human Health Risk Assessment and Ecological Risk Assessment, TNT Areas A and C***, Final, Plum Brook Ordnance Works, Sandusky, Ohio, November.

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Shaw Environmental, Inc. (Shaw), 2005, ***2004 Groundwater Data Summary and Evaluation Report***, Final, Former Plum Brook Ordnance Works, Sandusky, Ohio, April.

U.S. Environmental Protection Agency (EPA), 2004a, ***Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual (Part E - Supplemental Guidance for Dermal Risk Assessment)***, Final, Office of Superfund Remediation and Technology Innovation, Washington, D.C., EPA/540/R-99/005, July.

U.S. Environmental Protection Agency (EPA), 2004b, ***Region 9 PRG Table***, San Francisco, California, October.

U.S. Environmental Protection Agency (EPA), 2002, ***Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites***, Office of Solid Waste and Emergency Response, Washington, DC, 9355.4-24, December.

U.S. Environmental Protection Agency (EPA), 1997, ***Exposure Factors Handbook***, Office of Research and Development, National Center for Environmental Assessment, Washington, DC, August, EPA/600/P-95/002Fa.

U.S. Environmental Protection Agency (EPA), 1991, ***Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual (Part B, Development of Risk-based Preliminary Remediation Goals***, Interim, Office of Emergency and Remedial Response, December, EPA/540/R-92/003.

U.S. Environmental Protection Agency (EPA), 1989a, ***Risk Assessment Guidance for Superfund***, Volume I, Human Health Evaluation Manual (Part A), Interim Final, Office of Emergency and Remedial Response, Washington, D.C., EPA/540/1-89/002.