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June 25, 2013

U.S. Army Engineer District, Nashville
ATTN: CELRN-EC-E (Ms. Paula Coleman)
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Subject: ***Submittal of the Final Baseline Human Health Risk Assessment and Screening-Level Ecological Risk Assessment Work Plans for Waste Water Treatment Plant No. 2 (Waste Water Treatment Plant No. 2, TNT Area C to Waste Water Treatment Plant No. 2 Sewer Lines, and the Steel Sewer Line), Former Plum Brook Ordnance Works, Sandusky, Ohio Contract No. W912QR-08-D-0013: Shaw Project Number 140606***

Dear Ms. Coleman:

In accordance with the requirements of Delivery Order No. DX19 of Contract No. W912QR-08-D-0013 awarded to Shaw Environmental & Infrastructure, Inc., a CB&I company, we are pleased to submit the Final Baseline Human Health Risk Assessment and Screening-Level Ecological Risk Assessment Work Plans for the Waste Water Treatment Plant No. 2 (Waste Water Treatment Plant No. 2, TNT Area C to Waste Water Treatment Plant No. 2 Sewer Lines, and Steel Sewer Lines) at the Former Plum Brook Ordnance Works (PBOW) located in Sandusky, Ohio. This report was prepared consistent with other PBOW work plans and the agreements among the PBOW project delivery team.

Enclosed for your records are four copies of this report. Copies have also been sent to those on the distribution list as indicated for their records. As requested, the document was sent to the Center of Expertise (CX) and the Restoration Advisory Board Co-Chair in electronic format only.

Should you have any questions or require additional information regarding this submittal, please do not hesitate to contact me at (865) 694-7496.

Sincerely,

Steven. T. Downey, PE, PMP
Project Manager

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DISCIPLINE SIGN-OFF REVIEW

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Client Name: U.S. Army Engineer District, Nashville; CELRN-EC-E

Project Description: Risk Assessments for Waste Water Treatment Plant No. 2 (WWTP2, TNTC to WWTP2 Sewer Lines, and Steel Sewer Line) Remedial Investigation
Former Plum Brook Ordnance Works, Sandusky, Ohio

Contract No. W912QR-08-D-0013 Delivery Order No. DX19

Project No. 140606 Task/Phase Number: 12000000

Document Type		Identify specific section or segment covered by this checkprint	Document Origin
☐	Technical / Cost Proposal		X Originator Developed
☐	RFP		☐ Edited Standard
☐	Contract / Subcontract		☐ Client Furnished
X	Work Plans	Work Plans	
☐	Report		**Document Status**
☐	Risk Assessment / Evaluation		☐ Preliminary
☐	Specifications & Plans		☐ Internal Draft
☐	Design Calculations		☐ Draft
☐	Tables		☐ Draft Final
☐	Drawings / Figures		X Final
☐	Other:		☐ Other:

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NOTICE: By signature above, parties certify that the subject document has been prepared by and/or reviewed by them (as appropriate), that all review comments have been resolved, and that the document is ready for submittal.

Final
Baseline Human Health Risk Assessment Work Plan
Waste Water Treatment Plant No. 2,
TNT Area C to Waste Water
Treatment Plant No. 2 Sewer Lines, and
Waste Water Treatment Plants Steel Sewer Line
Plum Brook Ordnance Works
Sandusky, Ohio
FUDS Project No. G05OH001818

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Shaw Project Number 140606

June 2013

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List of Acronyms

α	attenuation coefficient of volatile organic compound concentration between soil gas and indoor air
ABS	dermal absorption factor
AF	soil-to-skin adherence factor
atm-m^3/mole	atmospheres per cubic meter per mole
bgs	below ground surface
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
cm^3/second	cubic centimeters per second
cm/hr	centimeters per hour
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
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
FUDS	Formerly Used Defense Sites

List of Acronyms *(Continued)*

g/cm-second	grams per centimeter per second
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
IEUBK	Integrated Exposure Uptake Biokinetic
ILCR	incremental lifetime cancer risk
IRIS	Integrated Risk Information System
IT	IT Corporation
kg	kilogram
kg/day	kilograms per day
kg/m^3	kilograms per cubic meter
K_p	permeability coefficient
L/cm^3	liters per cubic centimeter
L/day	liters per day
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/cm^2-day	milligrams per square centimeter per day
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

List of Acronyms *(Continued)*

mg/L	milligrams per liter
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	regional screening level
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
TNTC/WWTP2 SL	TNT Area C to Waste Water Treatment Plant No. 2 sewer lines
UCL	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 No. 1
WWTP2	Waste Water Treatment Plant No. 2
WWTPs SSL	Waste Water Treatment Plants Steel Sewer Line
WWTP3	Waste Water Treatment Plant No. 3

1.0 Introduction

This baseline human health risk assessment (BHHRA) work plan was prepared to describe the protocol for evaluating potential human health risks of exposure to soil and overburden/shale groundwater associated with the following three areas : Waste Water Treatment Plant No. 2 (WWTP2), the trace of the waste water sewer lines which extended from the former TNT Area C (TNTC) to the former WWTP2 (referred to hereinafter as TNTC/WWTP2 SL), and the trace of the steel sewer line that originated at WWTP2 and connected the former waste water treatment plants (WWTP) (referred to hereinafter as the WWTPs SSL). These three areas are 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. WWTP2, the TNTC/WWTP2 SL, and the WWTPs SSL will be evaluated separately in the BHHRA.

This work plan is consistent with U.S. Environmental Protection Agency (EPA) guidance and with the procedures established in the BHHRA for TNT Area A (TNTA) and TNTC soil (IT Corporation [IT], 2001a), the BHHRA work plan for groundwater at PBOW (Shaw Environmental, Inc. [Shaw], 2005a), the BHHRA work plan for Waste Water Treatment Plant No. 1 (WWTP1) and Waste Water Treatment Plant No. 3 (WWTP3) (Shaw, 2009), and the BHHRA work plan for the sewer lines extending from TNTA to WWTP1 and from TNT Area B (TNTB) to WWTP1 (Shaw, 2010).

1.1 Facility Description and Location

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. Hunting is allowed by permit on portions of PBOW during the annual deer hunting season.

1.2 Facility History and Background

The PBOW facility was constructed on property comprising 9,009 acres in early 1941 as a manufacturing plant for 2,4,6-trinitrotoluene (TNT), 2,4-dinitroluene, and pentolite (International Consultants Incorporated, 1995). Production of explosives at PBOW began in December 1941 and continued until 1945. It is estimated that more than 1 billion pounds of nitroaromatic explosives were manufactured during the 4-year operating period. The three explosive manufacturing areas were designated TNTA, TNTB, and TNTC. Twelve process lines were used in the manufacture of TNT: four lines at TNTA, three lines at TNTB, and five lines at TNTC.

After plant operations ceased, the manufacturing process lines were decontaminated by the Army in late 1945. During decontamination, all structures, equipment, and manufacturing debris were either removed and salvaged or removed and burned. After the property was certified as decontaminated, 3,230 acres of the property were initially transferred to the Ordnance Department, then to the War Assets Administration. In 1949, PBOW was transferred to the General Services Administration. This transfer did not include the Plum Brook Depot area, which consists of 2,800 acres. The Department of the Army reacquired the 3,230 acres in 1954. In 1955, the Army completed further decontamination of the manufacturing process lines. This effort included removal of contaminated surface and subsurface soil around the building and wooden and ceramic waste disposal lines containing TNT. Thousands of pounds of TNT were discovered in catch basins; this TNT was removed and burned at the burning grounds. The Army continued cleanup efforts until 1963.

Two property use agreements were entered into by the Army and the National Advisory Committee for Aeronautics, the predecessor of the National Aeronautics and Space Administration (NASA), in 1956 and 1958, respectively. Accountability and custody for the entire portion of the former PBOW property (6,030 acres) that had been under the accountability and custody of the Department of the Army were transferred to NASA on March 15, 1963. NASA performed further decontamination efforts during 1964. The NASA decontamination process included removing contaminated surface soil above the drain tiles, flumes, etc.; destruction of all buildings by fire; and removal of all soil, debris, sumps, and above-grade portions of concrete foundations. Portions of the concrete foundations located below grade were left buried, and some that had been previously slightly above grade were covered with fill material. All materials, including the soil in those areas, were flashed; the area was then rough-graded. The decontamination process was also to have included the burning of excavated nitroaromatic-filled pipelines (Dames & Moore, Inc. [D&M], 1997).

NASA has operated and maintained the former PBOW property since 1963, and the facility is currently the NASA Glenn Research Center, Plum Brook Station. NASA operates the property as a space research facility in support of their John H. Glenn Research Center at Lewis Field, Cleveland, Ohio. Most of the aerospace testing facilities built in the 1960s at PBOW are currently on standby or inactive status. On April 18, 1978, NASA declared approximately 2,152 acres of PBOW as excess. This excess included former buffer areas that had not been used by the Army and were thus not subject to decontamination efforts. The Perkins Township Board of Education acquired 46 acres of the excess acreage and uses this area as a bus transportation area. The General Services Administration retains ownership of the remaining excess acreage and currently has a use agreement with the Ohio National Guard for 604 acres of this land. The details of land transactions are listed in the site management plan (International Consultants Incorporated, 1995).

1.3 WWTP2, TNTC/WWTP2 SL, and WWTPs SSL Description and History

During production in the 1940s, three WWTPs were used to process production waste water from the three TNT manufacturing areas at PBOW. The wastes were accumulated in the settling basins of the TNT manufacturing areas. These wastes were transported to the WWTPs via aboveground and below-ground wood-stave sewer lines (USACE, 1995). Wood-stave pipes were constructed of small wood slats (i.e., staves) joined together in a tongue-and-groove fashion and reinforced with steel banding. Use of wood-stave pipes was not uncommon for water and sewage conveyance during the late 1800s until the 1950s.

Three WWTPs (WWTP1, WWTP2, and WWTP3) at PBOW received waste water from TNT manufacturing operations (Figure 1-2). WWTP1 is located nearest to TNTA and received waste water from the settling basins at TNTA and from TNTB. WWTP2 received waste water from TNTC via the wood-stave TNTC/WWTP2 SL and is located northeast of the West Area Red Water Pond, east of Acid Area No. 2, and north of TNTC. WWTP3 is located between WWTP1 and WWTP2 and northwest of TNT Area B. This work plan addresses risks associated only with WWTP2, the TNTC/WWTP2 SL, and the WWTPs SSL. WWTP2 is approximately 1.4 acres in size. The lateral extent of influence was assumed to be 10 feet to either side of the sewer line trace for both the TNTC/WWTP2 SL and the WWTPs SSL, resulting in approximate site sizes of 0.7 acre and 4.6 acres, respectively. Risks associated with the other WWTPs and their corresponding sewer lines are being addressed under DERP-FUDS Project No. G05OH001817.

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 the Pentolite Road Red Water Pond.

The waste water that was received from 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, nitrotoulenes, and dinitrotoluenes (D&M, 1996).

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 liquor storage tank. WWTP1 and WWTP2 also included a caustic storage tank and a raw waste storage tank. Both WWTP1 and WWTP2 received waste water from the TNT manufacturing settling basins and neutralized the slurry through a chemical depuration process. The liquid was thickened by evaporation and then incinerated. The incinerator was located south and west of the storage tanks at WWTP2 (Figure 1-2). Ash from the incinerator was disposed of in the nearby Power House 2 Ash Pits (USACE, 2000). Although incineration is believed to be the main treatment process for the waste water, historical design drawings suggest waste water may have been discharged directly to nearby ponds during periods of high production (e.g., the West Area Red Water Pond and Pentolite Road Red Water Pond). This is based on historical drawings showing the presence of a waste water line leading from the raw waste storage tank to the West Area Red Water Pond.

The WWTPs SSL was constructed as a 4-inch-diameter steel sewer line. This sewer line connected WWTP1 and WWTP2 to WWTP3. The layout of WWTP3 was similar to the other WWTPs except that it did not have caustic storage or raw waste water storage tanks. The lack of the caustic storage and raw waste water storage tanks at WWTP3 and the lack of sewer lines directly connecting WWTP3 with any of the TNT manufacturing areas suggest it was used to treat neutralized waste from WWTP1 and WWTP2. The steel sewer line was connected to the evaporator buildings at all three WWTPs and is interpreted to have been used to transfer neutralized waste to WWTP3 during periods of high productivity. Under a previous delivery order, a geophysical survey was conducted at approximately 10 locations along the sewer line. The purpose of the geophysical survey was to determine if the steel sewer line was still present. Results of the geophysical survey indicate the steel sewer line is present and is located at an estimated depth of 3 to 5 feet below ground surface (bgs). Subsequent trenching activities verified the presence of the steel sewer line.

During PBOW operations, the sewer lines reportedly often became clogged with TNT residue, and in some instances were completely plugged. The plugged lines were abandoned, and larger-diameter bypass sewer lines were constructed around the blocked areas to provide continual

drainage of the waste water (USACE, 1995). It is unknown whether clogging was an issue or if bypass lines were constructed for either of the sewer lines (TNTC/WWTP2 SL or WWTPs SSL) that are addressed by this work plan.

1.4 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. Approximately 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 overburden groundwater system exists within the unconsolidated material atop the bedrock under much of the site. This overburden system is hydraulically connected to the shale and is referred to as the overburden/shale groundwater unit (Section 3.1.1).

Current use of the PBOW facility is classified as industrial for the purpose of identifying plausible human receptors and exposure pathways for evaluation in the BHHRAs. Land use adjacent to the PBOW facility is primarily rural residential/agricultural, and residential development is a likely use if NASA were to excess property that includes the three sites under the WWTP2 investigation. D&M (1997) describes potential future uses of all or portions of the facility as follows:

- Industrial use may be continued (NASA activities and programs).
- Portions of the site may be used by hunters and fishermen for recreation.
- 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 WWTP2 and the property traversed by the TNTC/WWTP2 SL and the WWTPs SSL is considered to be industrial or residential for the purpose of developing receptor and exposure scenarios. Hunting is permitted in each of these areas; therefore, future use

of this property for hunting is evaluated in the BHHRAs. It is preliminarily assumed that overburden/shale groundwater may be developed as a source of potable water in the future. However, at most other PBOW sites, the volume of groundwater yielded by the overburden/shale unit is insufficient for use as a potable source, and the groundwater does not meet drinking water quality criteria because of natural contaminants in the water. Section 3.1.3 provides a more detailed discussion of receptors and exposure scenarios.

1.5 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 property along the former WWTP2, TNTC/WWTP2 SL, and WWTPs SSL traces. This work plan is intended to serve as the template for the BHHRAs report. Each BHHRA is a stand-alone document, chapter, or section; for example, 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 Ohio Environmental Protection Agency (OEPA) guidance, including, but not limited to, the following:

- Ohio Environmental Protection Agency (OEPA), 2009a***, Use of U.S. EPA's Regional Screening Levels as Screening Values in Human Health Risk Assessments***, Technical Decision Compendium, Division of Emergency and Remedial Response, August.

- Ohio Environmental Protection Agency (OEPA), 2009b***, Human Health Cumulative Carcinogenic Risk and Non-carcinogenic Hazard Goals for the DERR Remedial Response Program***, Technical Decision Compendium, Division of Emergency and Remedial Response, August.

- 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), 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.

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

- 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), 2010a, ***ProUCL Version 4.1 Technical Guide***, Draft, Office of Research and Development, Technology Support Center Characterization and Monitoring Branch, Las Vegas, Nevada, EPA/600/R-07/041, May.

- U.S. Environmental Protection Agency (EPA), 2010b, ***ProUCL Version 4.1 User Guide***, Draft, Office of Research and Development, Technology Support Center Characterization and Monitoring Branch, Las Vegas, Nevada, EPA/600/R-07/038, May.

- U.S. Environmental Protection Agency (EPA), 2011, ***ProUCL Version 4.1.01***, 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.

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 differences represent refinements or updates, 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 performed using the present protocol.

Ideally, this work plan captures and solidifies all details of the protocol for a BHHRA regarding the associated property. However, human health risk assessment knowledge and protocol are dynamic, and improvements and refinements may 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 BHHRAs and the cost of changing the protocol.

1.6 Work Plan Organization

The remainder of this document describes the components of the BHHRA process and 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. It is noted that the background screening protocol differs from the current OEPA (2009c) guidance, as explained in Section 2.4.3.

- ***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 BHHRAs, 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 BHHRAs and discusses their individual effects on the risk assessment results, focusing on those issues that are most pertinent to WWTP2 and the sewer line traces and/or those most likely to have the greatest effect on risk estimates.

- ***Chapter 7.0, Development of Risk-Based Remediation Criteria.*** Provides risk-based remediation criteria (RBRC) based on the methodology of the BHHRAs. 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 directly relevant to the risk assessment conclusions for WWTP2, the TNTC/WWTP2 SL, and the WWTPs SSL that are 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 each of the environmental media. WWTP2 and the TNTC/WWTP2 SL and WWTPs SSL traces will be evaluated as three separate sites.

2.1 Data Sources

All analytical data used in the BHHRAs 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, and monitoring well samples, as applicable. The sample summary table will identify each sample and the associated analytical suite used for each site. This will include the remedial investigation (RI) samples described in the RI work plan (Shaw, 2011).

2.2 Organization of the Analytical Data

Prior to initiation of BHHRA calculations, a database of chemicals present in site samples will be compiled separately for each site and medium. This database includes all chemicals detected. Surface 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 BHHRAs. 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.

Surface soil is defined as samples taken from 0 to 1 foot bgs, and subsurface soil is defined as samples taken from depths greater than 1 foot bgs. As described in the RI work plan (Shaw, 2011), subsurface soil samples included direct-push samples collected at a beginning depth greater than 1 foot and test pit soil samples collected immediately below the sewer line.

2.3 Evaluation of Data Quality

The quality of the analytical data will be evaluated to select data for inclusion in the BHHRAs. 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 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 BHHRAs as detected concentrations; "R" data and "B" qualified organic chemical data are not. "U" qualified data (nondetects) are treated in the BHHRAs 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 BHHRAs. 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 BHHRAs.

Some chemicals may be analyzed under two different analytical programs. For example, the dinitrotoluene isomers are analyzed by EPA Method 8330 for nitroaromatics as well as by 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 BHHRAs, 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 chemical 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 that 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) of a chemical in a given medium is compared to the appropriate risk-based screening concentration (RBSC) for that chemical and medium. This is performed for each chemical in each medium. 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 is not considered further in the BHHRA for this medium 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 BHHRAs are derived from the EPA (2012a) regional screening level (RSL) table "tap water" values, and RBSCs for soil are derived from "residential soil" RSL values. This is a change in the source of the RBSCs from PBOW BHHRA work plans begun prior to March 2009 based on discussion between USACE and OEPA (2009d), and this change is consistent with recent OEPA (2009a) guidelines. Previously, the groundwater and soil RBSCs were derived from the corresponding EPA (2004b) Region 9 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 BHHRAs, 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 (OEPA, 2009a; EPA, 2012b). This adjustment results in RBSC values associated with an HQ of 0.1. For cancer risk, the RSL values based on

an ILCR of 1E-6 were used directly as RBSCs in the BHHRAs. 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. OEPA recognizes an overall cancer risk of 1E-5, which represents the logarithmic midpoint of the EPA risk management range, as a remedial goal (OEPA, 2009b). 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 screening of lead in soil and groundwater is a special case. The EPA (2012c) Office of Water treatment technique action level of 15 µg/L for lead is listed in the RSL table, and the RSL user's guide recommends this level for use as an RSL (EPA, 2012b). Lead exposure and risk is evaluated separately from other chemicals using the EPA (2004c) Integrated Exposure Uptake Biokinetic (IEUBK) model. The selection of the action level as the drinking water RSL is based partly on the IEUBK model. Section 5.2 of the RSL user's guide states that if the average tap water concentration exceeds 15 µg/L and the average soil concentration exceeds a value of 250 milligrams per kilogram (mg/kg), then more than the IEUBK target (EPA, 2004c) of 5 percent of the population of exposed children would exceed 10 micrograms per deciliter of lead in blood. Because the Office of Drinking Water action level of 15 µg/L can be used to conservatively screen for a potential average concentration of 15 µg/L, this concentration is used as the RBSC.. However, it is possible that the residential soil RSL of 400 mg/kg, which is selected as the soil RBSC, may not screen out lead as a COPC when the average soil concentration exceeds 250 mg/kg within a given data set. Therefore, the following conditions were placed on the screening of lead: 1) If either the soil RBSC or groundwater RBSC is exceeded, then the IEUBK blood-lead model is run using both average soil and groundwater concentrations, and 2) if the average soil concentration exceeds 250 mg/kg, then the IEUBK model is run, even if neither RBSC is exceeded, using average concentrations of lead in both soil and groundwater.

Sulfate is another chemical that requires special consideration. There are no RSLs for sulfate in tap water, but the drinking water outside-of-the-mixing-zone average value of 250 milligrams per liter (mg/L) for the Lake Erie Basin is used as the RBSC. This value is the same as the secondary drinking water regulation of 250 mg/L (EPA, 2012c). Secondary drinking water regulations are nonpromulgated values, based on aesthetic characteristics, which are used as guidelines for public water systems. A health-based advisory level of 500 mg/L also exists for sulfate (EPA, 2012c).

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. No RSLs are listed for these nutrients. Should any of these chemicals be identified as site related, an exposure analysis is performed whereby a daily dose of chemical from ingestion of the medium in question is calculated. The dose is 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 BHHRAs. 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 BHHRAs, chemicals that are reported 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 reported at a high concentration or 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-U.S. 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 (Section 3.1.1).

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 (Section 2.4.3.1) and is performed after the risk characterization (Chapter 5.0). The results of the statistical data testing are discussed in the uncertainty analysis (Chapter 6.0). No suitable background data set exists for overburden/shale

groundwater, so no background screening or statistical comparisons to background concentrations can be made for overburden/shale groundwater samples. Please note that the only groundwater samples that were collected for WWTP2, the TNTC/WWTP2 SL, and the WWTPs SSL were from the overburden/shale groundwater. Therefore, background screening will not apply to groundwater samples from these three sites.

Inorganics and organics are 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) are carried through the risk calculation process (exposure assessment, toxicity assessment, and risk characterization). As presented in Section 2.4.3.3, organic compounds are quantitatively eliminated as background related only through the uncertainty analysis if applicable.

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 bedrock groundwater are described in the 2004 groundwater report (Shaw, 2005b). It is noted that the method agreed upon for the development of BSCs, as recorded in the September 11, 2002 PBOW team meeting minutes, differs from that shown in current OEPA (2009c) guidance. This PBOW team background screening method, which has been used for all PBOW risk assessments over a period of more than a decade, will be used for the WWTP2 and associated sewer line BHHRAs to ensure consistency in the evaluation between PBOW sites. 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 may be regarded as a COPC if its MDC exceeds the BSC for that chemical or if no BSC can 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 presented in the 2004 groundwater report (Shaw, 2005b), and the background data set for soil is presented in the site investigation for the acid areas (IT, 1998). As mentioned previously, background data sets do not exist for overburden/shale groundwater; therefore, a statistical background evaluation for COPCs in this medium 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 are appended to the BHHRAs for each inorganic data set evaluated against the appropriate site background data set; the WRS results are discussed as part of the uncertainties. 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 is not run 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).

2.4.3.3 Treatment of Organic Compounds

As mentioned in Section 2.4.3, certain organic compounds (e.g., BTEX and PAHs) in site media may be attributable to background conditions. However, no organic compounds are summarily screened out. Instead, all detected organic compounds are carried through the risk assessment process (i.e., exposure assessment, toxicity assessment, and 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 contributions of organics are discussed in the uncertainties analysis, as applicable.

2.5 Data Evaluation Summary

Tables with the following information for each detected chemical in each environmental medium will be included in the BHHRAs:

- 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) (for COPCs only)
- Appropriate RBSC
- Appropriate BSC
- Selection/exclusion of chemical as a COPC
- EPC (for COPCs only).

Separate sets of tables will be prepared for WWTP2, the TNTC/WWTP2 SL, and the WWTPs SSL. 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 BHHRAs. 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 piezometer samples will be appended to the BHHRAs. Note that these direct-push groundwater samples are collected for nature-and-extent purposes to determine groundwater flow direction and the placement of 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 BHHRAs. These piezometer 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 BHHRAs described in this work plan will characterize potential exposures to COPCs in soil and groundwater associated with WWTP2, the TNTC/WWTP2 SL, and the WWTPs SSL, as portrayed by the CSEM in Section 3.1.

3.1 Conceptual Site Exposure Model

The CSEM provides the basis for identifying and evaluating the potential risks to human health in the BHHRAs. The CSEM, graphically depicted on Figures 3-1 and 3-2 for WWTP2, the TNTC/WWTP2 SL, and the WWTPs SSL, respectively, includes the receptors appropriate to all plausible site use scenarios and the potential exposure pathways. 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 BHHRAs will state that the site-specific risk assessment, including the evaluation of future land use and groundwater use, was performed to satisfy administrative requirements. 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 for WWTP2, the TNTC/WWTP2 SL, and the WWTPs SSL that were 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). Figure 3-2 depicts the receptors and pathways for WWTP2, the TNTC/WWTP2, and the WWTPs SSL. Asterisks on these figures identify exposure pathways that are complete and will be addressed in the BHHRAs. Justification for exclusion of other pathways is provided in the figure footnotes, and the exclusion of other receptors is discussed in Section 3.1.3.7. No current or future exposure 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 are used as potable sources. 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 [2008]); however, the groundwater underlying these sites is not 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 the overburden/shale groundwater is not plausible, that information will be presented in the BHHRA reports.

3.1.1 Physical Setting

The topography in the vicinity of the WWTP2 is generally flat, sloping southeast toward Pipe Creek. Elevation differences across the site are a total of approximately 18 feet. The TNTC/WWTP2 SL trace is essentially flat south to north, sloping downward approximately 14 feet toward Pipe Creek. The intersection of the sewer line trace with Pipe Creek is located approximately 500 feet from the northern terminus of trace. The WWTPs SSL traverses the central portion of PBOW from the WWTP1 site on the eastern side of the facility to WWTP2 on the west-central side. The trace of the WWTPs SSL is relatively flat, varying only about 14 feet in elevation throughout its 10,000-foot length. Its lowest point is at its crossing of Pipe Creek.

There are currently no structures above grade at WWTP2, although the area is littered with building demolition debris including concrete blocks, drainage tiles, and rebar. There are no above-grade structures along the trace of the TNTC/WWTP2 SL or the WWTPs SSL.

WWTP2 is currently forested, with a mown area southwest of the incineration building. Vegetation along the TNTC/WWTP2 SL includes forested and shrubby areas. Vegetation along the WWTPs SSL varies from short shrubs, grasses, and forbs in the east to west-central parts of the trace to mature trees and shrubs on the western side near WWTP2.

The following subsections further describe the physical setting of WWTP2, the TNTC/WWTP2 SL, and the WWTPs SSL, including the climate and meteorology of the Sandusky region (Section 3.1.1.1), geology (Section 3.1.1.2), soil (3.1.1.3), hydrology (Section 3.1.1.4), and surface water characteristics (Section 3.1.1.5) of the three sites. A general description of the WWTP2, the TNTC/WWTP2, and the WWTPs SSL is provided in Section 1.3.

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 three per month), and the last freezing day falls in early May (average of one 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 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).

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 PBOW facility 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). The shale is typically thin to absent in the western part of PBOW.

Based upon the location of WWTP2 within the regional geologic bedrock interpretation, the underlying bedrock at the WWTP2 area is believed to be almost entirely limestone (Delaware Limestone). Shale was encountered in only 1 (PZ01) of the 14 borings drilled during the WWTP2 investigation, and no borings were drilled into the bedrock (Shaw, 2012). Based upon the location of TNTC/WWTP2 SL within the regional geologic bedrock, the underlying bedrock of the sewer line traverses both the Olentangy Shale and the Delaware Limestone. Thickness of the shale at the southern terminus of the TNTC/WWTP2 SL is estimated, based on the nearest TNTC well boring (PB-BED-MW13), to be approximately 29 feet thick (Shaw, 2012). The shale likely thins gradually south to north along the sewer line trace to its northern terminus at WWTP2, where the shale is thin to absent. Based on bedrock borings from adjacent PBOW sites, the shale is likely to be thickest in the eastern portion of the WWTPs SSL, where it is estimated to be 14 feet thick. It likely thins to approximately 2 feet in the west-central portion of the trace near the Reservoir No. 2 Burning Ground (Figure 1-2) and is virtually nonexistent at its western terminus at WWTP2.

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 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 vicinity of WWTP2, the thickness of the overburden was found to range from 7 to 16 feet. The thickness of the overburden along the TNTC/WWTP2 SL ranged from 11 to 21 feet, based on soil boring logs. In the vicinity of the WWTPs SSL, the overburden thickness was found to range from approximately 20 to 27 feet.

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, 1997, 1999, 2001b) 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 sewer line traces. 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 a localized groundwater flow direction that is likely to be to the northwest in the vicinity of the sewer line traces.

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). WWTP2 is in Zone 1; the TNTC/WWTP2 SL is in Zones 1 and 3; and the WWTPs SSL is mostly in Zone 3, with portions in the other two zones.

Groundwater was encountered at WWTP2 in only 5 of 14 borings, and at depths ranging from 5.0 to 12.5 feet bgs. Along the TNTC/WWTP2 SL, groundwater was encountered in only 2 of 13 borings; at one of these, groundwater was encountered in the overburden at a depth of 8.4 feet, and in the other, groundwater was encountered in a perched sand lens found at a depth of 4 to 8 feet bgs. Along the WWTPs SSL, groundwater was encountered at three of the six borings, with depth to groundwater ranging from 2 to 11 feet among these three borings. The shallowest boring was in the eastern part of the trace, near WWTP1, and the deepest was in the western portion of the WWTPs SSL, near WWTP2. No limestone wells have been installed at any of these three areas.

Surface Water. The TNTC/WWTPs SL crosses Pipe Creek, south of WWTP2. Note that surface water and sediment samples from this area have been included and evaluated under other PBOW projects and are not evaluated again for these three sites. The WWTPs SSL crosses Pipe Creek, Ransom Brook, a tributary to Pipe Creek east of the Garage Maintenance Area, and numerous small, ephemeral ditches along its length, as shown on Figure 1-2.

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). WWTP2 received waste water via sewer lines from the TNTC waste water settling basins. This waste water was either treated and evaporated (thickened to reduce mass) or the liquid was discharged to the West Area Red Water Ponds. If treated, the thickened liquid was incinerated, and the ash from this incineration was disposed of in Ash Pit 2. 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 carried contaminants to nearby streams. Infiltration and leaching carried contaminants into the subsurface soil or groundwater.

With respect to the sewer lines, releases would have occurred within the subsurface soil. The TNTC/WWTP2 SL appears to have been removed; therefore, some of the contaminated subsurface soil may have been mixed and brought to the surface during removal of the lines. The WWTPs SSL is still present. Therefore, the soil mixing that may have occurred at the TNTC/WWTP2 is not expected to have occurred at the WWTPs SSL.

It is possible that surface water and sediment at the drainage ditches may have been impacted by contaminants originating from WWTP2 and/or from the TNTC/WWTP2 SL and WWTPs SSL through surface drainage. However, the surface water and sediment near WWTP2 have been addressed under other PBOW projects and will not be re-evaluated in these BHHRAs. The WWTPs SSL also crosses numerous small drainage ditches (Figure 1-2). These have either been evaluated as part of other PBOW projects or are wet weather conveyances which overlie the sewer line. In this latter case, any contamination associated with leaks would be expected to migrate downward.

3.1.3 Receptors and Exposure Pathways

Receptors, selected to represent the upper bound on exposure from all groups of people plausibly exposed to environmental media associated with WWTP2 and the associated waste water sewer lines, as well as the pathways by which they may be exposed to chemicals, are summarized on Figures 3-1 and 3-2 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 (2005) policy of carcinogen evaluation, although it introduces considerable uncertainty into 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 or has unusual dimensions so that a groundskeeper would be unlikely to spend all of his working time at the site, an FI value of less than 1 might be applied to the soil ingestion and dermal intake equations.

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 because 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 to 10 feet bgs) will be spread as surface soil (0 to 1 foot bgs). Groundwater use is also evaluated for the groundskeeper in the future site-use scenario (discussed in Section 3.1), in which groundwater could theoretically be developed as a source of drinking water.

Direct soil exposure pathways include incidental ingestion and dermal contact. Inhalation of dust raised by operating lawnmowers 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 that is 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., the first few centimeters) from which volatilization would occur. However, as noted previously, the data set for surface soil may include samples taken from up to 1 foot 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 foot bgs) would be attenuated by the overlying soil so that concentrations in ambient air would not be toxicologically significant.

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 square centimeter (mg/cm^2) are assumed (EPA, 2004a). An FI term of 1 will be used for site soil exposure pathways in the initial exposure assessment. However, given the linear nature of the area of investigation along a sewer line trace, an FI value of less than 1 would certainly be more realistic for this receptor. Therefore, if appropriate, an alternative FI may be considered in the uncertainty analysis.

A future groundskeeper is assumed to be exposed to groundwater, which could theoretically be developed as a source of drinking water at some time 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 (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, direct contact with surface soil for the indoor worker is included to reflect a more complete exposure scenario, but direct contact of surface soil 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). An FI term of 1 will be used for exposure site soil pathways in the initial exposure assessment. However, given the linear nature of the area of investigation along a sewer line trace, a value of less than 1 would certainly be more realistic for this receptor. Therefore, if appropriate, an alternative FI may be considered in the uncertainty analysis of the BHHRAs.

A future indoor worker is assumed to be exposed to groundwater, which could theoretically be developed as a source of drinking water (Section 3.1). His drinking water ingestion rate is assumed to be 1 L/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 (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 is assumed to be a 70-kg adult who works 8 hours/day, approximately 5 days/week. This represents an annual exposure frequency rate of about 250 days per year. 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/hr) (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; 2004a). An FI term of 1 will be used for site soil exposure pathways in the initial exposure assessment. However, given the linear nature of the area of investigation along a sewer line trace, an FI value less than 1 might be more realistic for this receptor. Therefore, if appropriate, an alternative FI may be considered in the uncertainty analysis of the BHHRAs.

3.1.3.4 On-Site Resident

Land use adjacent to the PBOW facility is primarily rural residential/agricultural, and residential development is likely if NASA were to excess property that includes the three sites under the

WWTP2 investigation. The on-site residential scenario is created to evaluate the upper bound for long-term exposure to site soil, and to groundwater under the future land-use scenario. The on-site residential scenario is evaluated assuming a 30-year 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; 2012c).

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, if appropriate based on the RI data, that under future residential land use, the overburden/shale unit will be developed as a source of potable water (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 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, 2004d). 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).

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, 2012c). 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 20,000 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, 2004d) 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. WWTP2 and property along the sewer line traces provides habitat for deer and other wildlife. Deer hunting is allowed by permit in these areas seasonally and is assumed to be permitted in the future in these areas. 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 is not evaluated for the reasons previously explained for other receptors.

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 regarding the rate of venison ingestion were not located; 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 property that includes the sewer line traces. 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. The sewer line traces 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.

As discussed in Section 3.1, off-site use of groundwater will not be evaluated because nearby residents use municipal water from surface water sources as a potable source, and potential on-site users would be exposed to higher concentrations.

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 and groundwater 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 and groundwater 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 and Groundwater 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 use of a UCL for RME evaluation as a conservative estimate of the true mean exposure concentration.

The EPA (2010a,b; 2011) ProUCL (Version 4.1) 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 (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, 1989a), as such values unduly make distribution testing impossible for ProUCL (EPA, 2010b). 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 BHHRAs.

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 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, with 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 BHHRAs (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):

Eq. 3.1

$$C_a = (D)(C_s)(CF_1)$$

where:

C_a	=	contaminant concentration in air (milligrams per cubic meter [mg/m^3], calculated)
D	=	dust loading factor (grams of soil/m^3 of air)
C_s	=	contaminant concentration in soil (mg/kg)
CF_1	=	conversion factor (1E-3 kg 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 (PEF) 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 kilograms per cubic meter (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 [Section 3.1.1])
U_t	=	equivalent threshold value of wind speed at 7 meters (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 (2004e) 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 no structures currently exist at WWTP2 or along the TNTC/WWTP2 SL and WWTPs SSL traces.

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 (α)
- Combining C_{source} and α 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 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 air models described in Section 3.2.2, then from EPA (1997b). The calculations may be performed in the BHHRAs using the EPA (2004e) vapor intrusion model, which is adapted from of the Johnson and Ettinger (1991) model.

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, 2004e):

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 (atmosphere(s) per cubic meter per mole [atm-m^3/mole], Chemical specific)
C_s	=	contaminant concentration in soil (mg/kg)
P_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 $K_{oc} \bullet f_{oc}$)
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-[P_b/P_s])
P_s	=	soil particle density (2.65 g/cm^3, default [EPA, 1996], or site specific).

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 (2004e) 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 follows:

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 (square centimeters per second [cm^2/second], calculated)
D_a	=	diffusivity in air (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-θ_w)
n	=	total capillary zone soil porosity (0.43 unitless, default [EPA, 1996], or site specific, estimated as 1-[P_b/P_s])
P_b	=	dry soil bulk density (1.5 g/cm^3, default [EPA, 1996], or site specific)
P_s	=	soil particle density (2.65 g/cm^3, default [EPA, 1996], or site specific).
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 (atm-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 follows:

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 cubic centimeters per second [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 centimeters [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 (2004e) assumes that the only crack available for the entry of soil gas is a 0.1-cm-wide gap at the interface of the floor and foundation. As noted previously, 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 flow rate of soil gas into enclosed space is calculated as follows:

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 grams per centimeter per second [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 cm, 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

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

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 (4,920 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, 2004e). 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 in Ambient Air from Subsurface Soil

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 (atm-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} • 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 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).

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 by the WWTP2 area or sewer line traces; therefore, the fraction of total browse consumed from the area that may be contaminated along the traces is expected to be relatively small.

- Indirect food chain pathways may be significant for some metals and for those semivolatile organic compounds that persist in the environment and tend 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 (0.06, 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 BHHRAs. Toxicity profiles will be prepared for each of the COPCs evaluated in the BHHRAs. The toxicity profiles briefly describe the uses of the chemical, its physical properties, behavior in environmental media, biotransfer capability, and toxicity values.

WWTP2 and the sewer line traces are relatively small in comparison to the home range of a white-tailed deer. The area of WWTP2 is approximately 1.4 acres. The total length of the TNTC/WWTP2 SL traces is approximately 1,700 feet. If it is assumed that a swath of surface soil 20 feet wide has been impacted by the sewer line, this represents a total area of approximately 0.7 acre. The total length of the WWTPs SSL is approximately 10,000 feet. Assuming that a swath of 20 feet is impacted by this sewer line, this represents an area of approximately 4.6 acres. 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 acreage represented by the largest of these areas is approximately 3 percent of the white-tail deer home range. Therefore, an FI_r value of 0.03 will be used in the BHHRAs for each of these areas.

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):

$$C_p = (CF)(C_s)(B_p) \qquad \text{Eq. 3.20}$$

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 BHHRAs. 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 a 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. Receptor-specific exposure parameter values from Table 3-2 are used in the equations described in Sections 3.3.1 through 3.3.5.

3.3.1 Inhalation of COPCs in Air

The following equation is used to estimate the inhaled dose of a COPC in air (groundskeeper, construction worker, 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 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 a COPC in soil (groundskeeper, construction worker, resident, indoor worker, hunter) is estimated from the equation:

Eq. 3.23

$$I = \frac{(C)(FI)(IR)(EF)(ED)(CF)}{(BW)(AT)}$$

where:

I	=	ingested dose of COPC in soil (mg/kg-day, calculated)
C	=	concentration of COPC in soil (mg/kg)
FI	=	fraction of exposure attributed to site soil (unitless)
IR	=	soil ingestion rate (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 Water

The ingested dose of a COPC in groundwater (future groundskeeper, resident) is estimated from the following equation:

$$I_w = \frac{(C_w)(IR_w)(FI_w)(EF)(ED)}{(BW)(AT)} \qquad \text{Eq. 3.24}$$

where:

I_w	=	ingested dose of COPC in water (mg/kg-day, calculated)
C_w	=	concentration of COPC in water (mg/L)
IR_w	=	water ingestion rate (L/day)
FI_w	=	fraction of exposure attributed to site water (unitless)
EF_w	=	fraction of exposure attributed to site water exposure frequency (days/year)
ED_w	=	exposure duration (years)
BW	=	body weight (kg)
AT	=	averaging time (days).

3.3.4 Dermal Contact with COPCs in Soil or Groundwater

Unlike the methodologies for estimating inhaled or ingested doses of COPCs, 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 (milligrams per square centimeter per day [mg/cm^2-day])
SA = SA_s for soil, SA_{gw} for groundwater, = 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).

Dose absorbed per unit body surface area per day (DA) is calculated differently for dermal uptake from soil and from water. Dermal uptake of constituents from soil (groundskeeper, construction worker, resident, hunter) 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

$$DA = (C_s)(FI_s)(CF)(AF)(ABS)$$

where:

DA = dose absorbed per unit body surface area per day (mg/cm^2-day, calculated)
C = concentration of COPC in medium (mg/kg)
FI = fraction of exposure attributed to site medium (unitless)
CF = conversion factor (1E-6 kg/mg)
AF = soil- 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 BHHRAs.

Quantification of dermal uptake of constituents from groundwater (future groundskeeper, 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. 3.26). DA is calculated differently for inorganic and organic chemicals in water. For inorganic chemicals, DA is calculated from the following equation:

Eq. 3.27

$$DA = (C_w)(FI)(K_p)(ET_w)(CF)$$

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)
K_p	=	permeability coefficient (centimeters per hour [cm/hr])
ET_w	=	time of exposure (hours/day)
CF	=	conversion factor (1E-3 liters per cubic centimeter [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., 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)
FA	=	fraction absorbed from the water (unitless)
K_p	=	permeability coefficient (cm/hr)
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_w	=	time of contact (hour(s)/day).

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

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

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)
FA	=	fraction absorbed from the water (unitless)
K_p	=	permeability coefficient (cm/hr)
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_w	=	time of contact (hour(s)/day).
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/hr, 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 BHHRAs will be summarized in a table of the BHHRAs. The values will be documented in toxicity profiles appended to the BHHRAs.

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:

I_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 (EPA, 2012d). 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 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 (2005) 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.

IRIS (EPA, 2012d) 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 (micrograms per cubic meter $[\mu g/m^3])^{-1}$. 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 micrograms per milligram, and dividing the result by 20 m^3/day.

4.2 Evaluation of Noncarcinogenic Effects

Many chemicals, whether associated with carcinogenicity or not, 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 is considerably less than 10 percent of the expected life span.

IRIS (EPA, 2012d) 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 BHHRAs.

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 hazard indices (HI) 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 BHHRAs 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, 2012d) 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 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 BHHRAs for all of the COPCs evaluated in the BHHRAs. Summary toxicity information sufficient to support the risk calculations, including toxicity values, GAFs, target organs, and sources, will be provided in tables with the BHHRAs.

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):

$$ILCR = (CDI)\,(SF) \qquad \text{Eq. 5.1}$$

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, 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

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

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

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

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 (2009b) and will be used in the BHHRAs. 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):

$$HQ = I \ / \ RfD \qquad \text{Eq. 5.5}$$

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 5.5 is equivalent to the "I" or "DAD" terms (intake or dose) in Equations 3.22 through 3.24 and 3.31 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:

$$HI = HQ_1 + HQ_2 + \ldots HQ_i \qquad \text{Eq. 5.6}$$

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 (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 BHHRAs 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 BHHRAs 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 site being evaluated.

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 BHHRAs 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 and 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 BHHRAs. Consequently, the RBRCs are specific to site, source medium, receptor, and chemical. 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 BHHRAs 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 BHHRAs may be revised by the project delivery team during the FS process based on other site-specific considerations.

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:

$RBRC_{coc-R}$	=	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 non-cancer-based RBRCs will be derived for COCs for which both cancer and non-cancer-based toxicity values are available.

8.0 Summary and Conclusions

This section will briefly summarize the BHHRAs 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. The conclusions will note that the site-specific risk assessment, including the evaluation of future land and groundwater use, was performed to satisfy administrative requirements.

9.0 References

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TABLES

Table 3-1

Receptor/Exposure Scenarios
Waste Water Treatment Plant No. 2 and Associated Sewer Lines
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]		
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
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
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]		

Table 3-1

Receptor/Exposure Scenarios
Waste Water Treatment Plant No. 2 and Associated Sewer Lines
Plum Brook Ordnance Works, Sandusky, Ohio

Source Medium	Model	Exposure Medium	Exposure Route
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
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]		
Hunter's Child – Future			
Surface Soil	Not Quantified[c]		
	Not Quantified[c]		
	Biouptake	Venison	Venison Consumption
Subsurface Soil	Not Quantified[c]		

[a] Although theoretically complete, this pathway is not quantified, as explained in text.
[b] Although contact with this medium may be 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 regrading is assumed to take place.
[e] Even though the mixing of surface and subsurface soil described in footnote "d" 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 Plant No. 2 and Associated Sewer Lines
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[d]	250[f]	350[a]	NA	NA
Exposure duration (ED), years	25[a]	0.5[f]	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[a]	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]	1[c]	1[c]	1[c]

Table 3-2

Variables Used to Estimate Potential Chemical Intakes and Contact Rates for Receptors
Waste Water Treatment Plant No. 2 and Associated Sewer Lines
Plum Brook Ordnance Works, Sandusky, Ohio

Pathway Variable	Grounds-keeper	Construction Worker	On-Site Resident	Indoor Worker	Hunter and Hunter's Child
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]
Exposure frequency (EF), days/year	250[d]	250[a]	350[a]	250[a]	14[d]
Exposure duration (ED), years	25[a]	0.5[f]	Child: 6[a] Adult: 24[a]	25[a]	30[a]
Dermal Contact with Soil					
Fraction exposed to contaminated medium (FI_{so}), unitless	1[c]	1[c]	1[c]	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[g,h]
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[g,h]
Dermal absorption factor (ABS), unitless	csv	csv	csv	NA	csv
Exposure frequency (EF), days/year	250[d]	250[a]	350[a]	NA	14[h]
Exposure duration (ED), years	25[a]	0.5[f]	Child: 6[a] Adult: 24[a]	NA	30[a]
Inhalation of VOCs from Groundwater					
Exposure time (ET), hours/day	NA	NA	24[i]	NA	NA
Inhalation rate (IR_a), m^3/hour	NA	NA	Child: 0.416[e] Adult: 0.833[e]	NA	NA
Exposure frequency (EF), days/year	250[d]	NA	350[a]	250[a]	NA

Table 3-2

Variables Used to Estimate Potential Chemical Intakes and Contact Rates for Receptors
Waste Water Treatment Plant No. 2 and Associated Sewer Lines
Plum Brook Ordnance Works, Sandusky, Ohio

Pathway Variable	Grounds-keeper	Construction Worker	On-Site Resident	Indoor Worker	Hunter and Hunter's Child
Drinking Water Ingestion of Groundwater					
Fraction exposed to contaminated medium (Fl_{gw}), unitless	1[c]	NA	1[c]	1[c]	NA
Drinking water ingestion rate (IR_{gw}), L/day	1[d]	NA	Child: 1[e] Adult: 2[d]	1[d]	NA
Exposure frequency (EF), days/year	250[d]	NA	350[a]	250[a]	NA
Dermal Contact with Groundwater					
Fraction exposed to contaminated medium (Fl_{gw}), unitless	1[c]	NA	1[c]	1[c]	NA
Body surface area exposed to water (SA_{gw}), cm^2	3,300[g,j]	NA	Child: 6,600[k] Adult: 20,000[k]	3,300[k]	NA
Permeability coefficient (PC), cm/hour	csv	NA	csv	csv	NA
Exposure time (ET_{gw}), hours/day	1[c]	NA	Child: 0.333[k] Adult: 0.2[k]	1[c]	NA
Exposure frequency (EF), days/year	250[d]	NA	350[d]	250[a]	NA
Venison Consumption					
Venison ingestion rate (IR_v), kg/day	NA	NA	NA	NA	Child: 0.005[h] Adult: 0.013[l]
Exposure frequency (EF), days/year	NA	NA	NA	NA	350[a]
Exposure duration (ED), years	NA	NA	NA	NA	Child: 6[a] Adult: 30[m]

Table 3-2

Variables Used to Estimate Potential Chemical Intakes and Contact Rates for Receptors
Waste Water Treatment Plant No. 2 and Associated Sewer Lines
Plum Brook Ordnance Works, Sandusky, Ohio

Note: Groundwater pathways are quantified only for Ash Pit 1 because no groundwater samples could be collected at Ash Pit 3 (see Section 2.1 of text).

[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] Default value which assumes 100 percent of exposure is to contaminated medium.

[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/region9/superfund/prg/files/04usersguide.pdf.

[f] Assumed; see Section 3.1.3.3 of text for explanation.

[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] Please see Section 3.1.3.6 of text for detailed explanation.

[i] The *Exposure Factors Handbook* (see reference g) indicates that the 90th percentile for the amount of time spent at a residence is more than 23 hours per day.

[j] Please see Section 3.1.3.1 of text for detailed explanation.

[k] U.S. Environmental Protection Agency (EPA), 1997, *Exposure Factors Handbook*, Final, National Center for Environmental Assessment, Washington, D.C., EPA/600/P-95/002Fa, August.

[l] Please see Section 3.1.3.5 of the text for detailed explanation.

[m] Assumes deer are hunted for a 30-year duration.

csv – Chemical-specific value.
NA – Pathway not applicable for receptor.

FIGURES

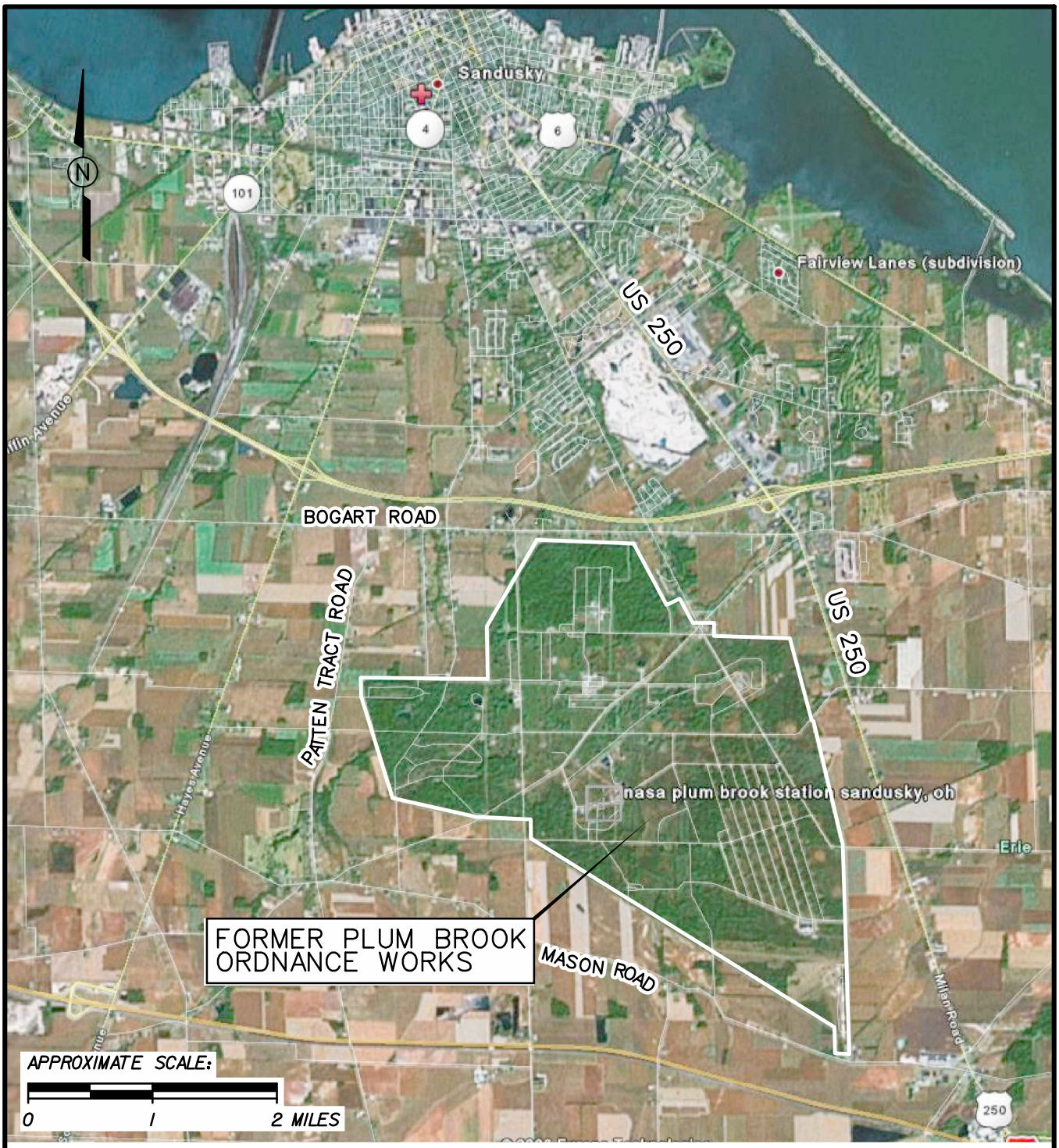

FIGURE 1-1
PBOW VICINITY MAP

WWTP2, TNTC/WWTP2 SL, AND WWTP2 SSL BASELINE HUMAN HEALTH RISK ASSESSMENT WORK PLAN
FORMER PLUM BROOK ORDNANCE WORKS
NASA PLUM BROOK STATION
SANDUSKY, OHIO

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wwtp2_tntcsl_ssl_blhhra_wp_001.dgn cbentley 7/17/2012 2:11:41 PM pdf_with_levels.plt SEI_TEXTSUB_ONLY.TBL

NOTES:

1. WWTP2 = WASTE WATER TREATMENT PLANT No.2.
2. TNTC/WWTP2 SL = TNT AREA C TO WASTE WATER TREATMENT PLANT No.2 SEWER LINES.
3. WWTP2 SSL = WASTE WATER TREATMENT PLANT No.2 STEEL SEWER LINES.

LEGEND:

- AREA OF CONCERN
- APPROXIMATE LOCATION OF SEWER LINES
- POND
- CREEK, DITCH, CONVEYANCE
- RAILROAD
- ROAD
- FENCE
- FACILITY BOUNDARY

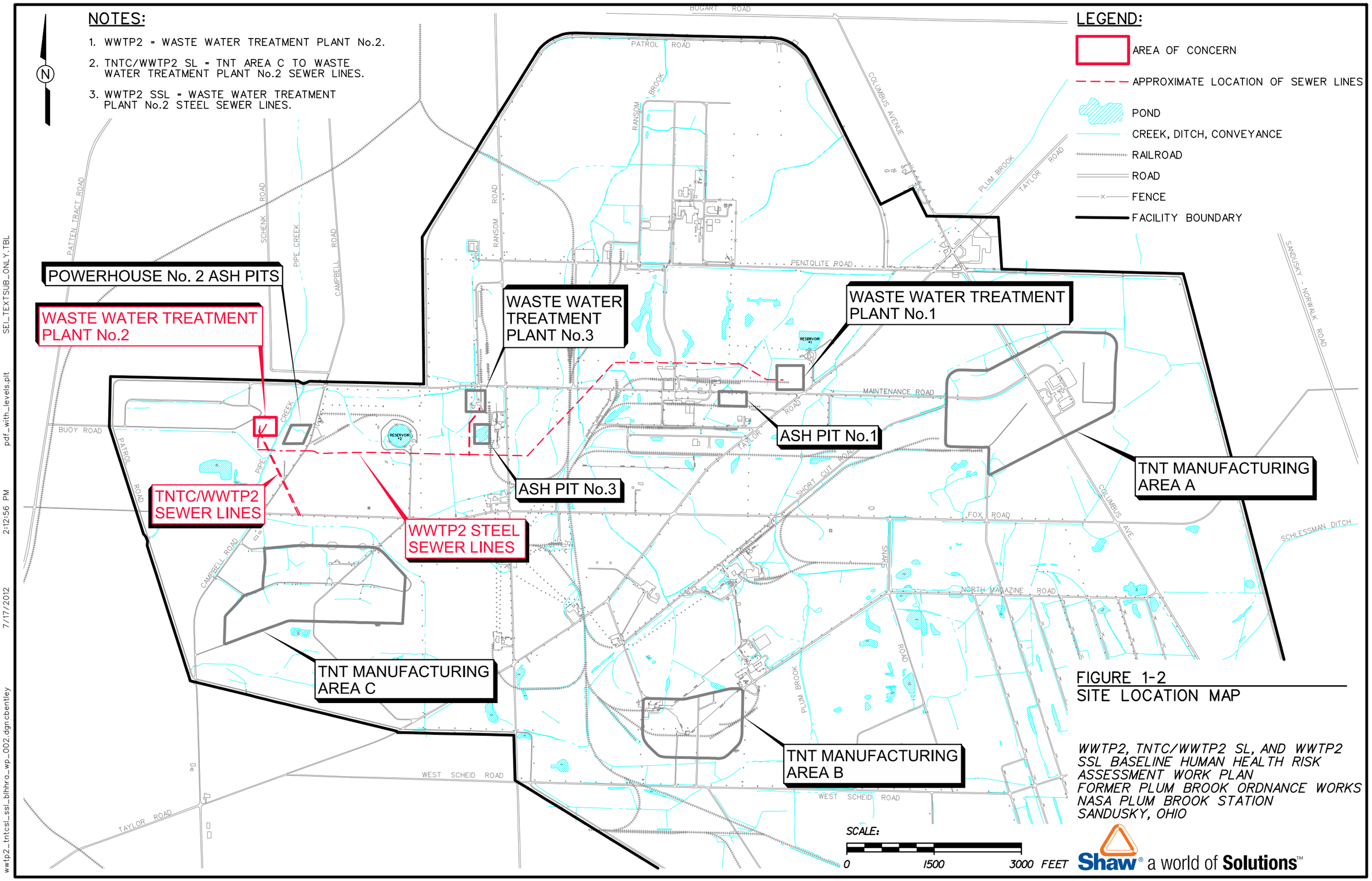

FIGURE 1-2
SITE LOCATION MAP

WWTP2, TNTC/WWTP2 SL, AND WWTP2 SSL BASELINE HUMAN HEALTH RISK ASSESSMENT WORK PLAN
FORMER PLUM BROOK ORDNANCE WORKS
NASA PLUM BROOK STATION
SANDUSKY, OHIO

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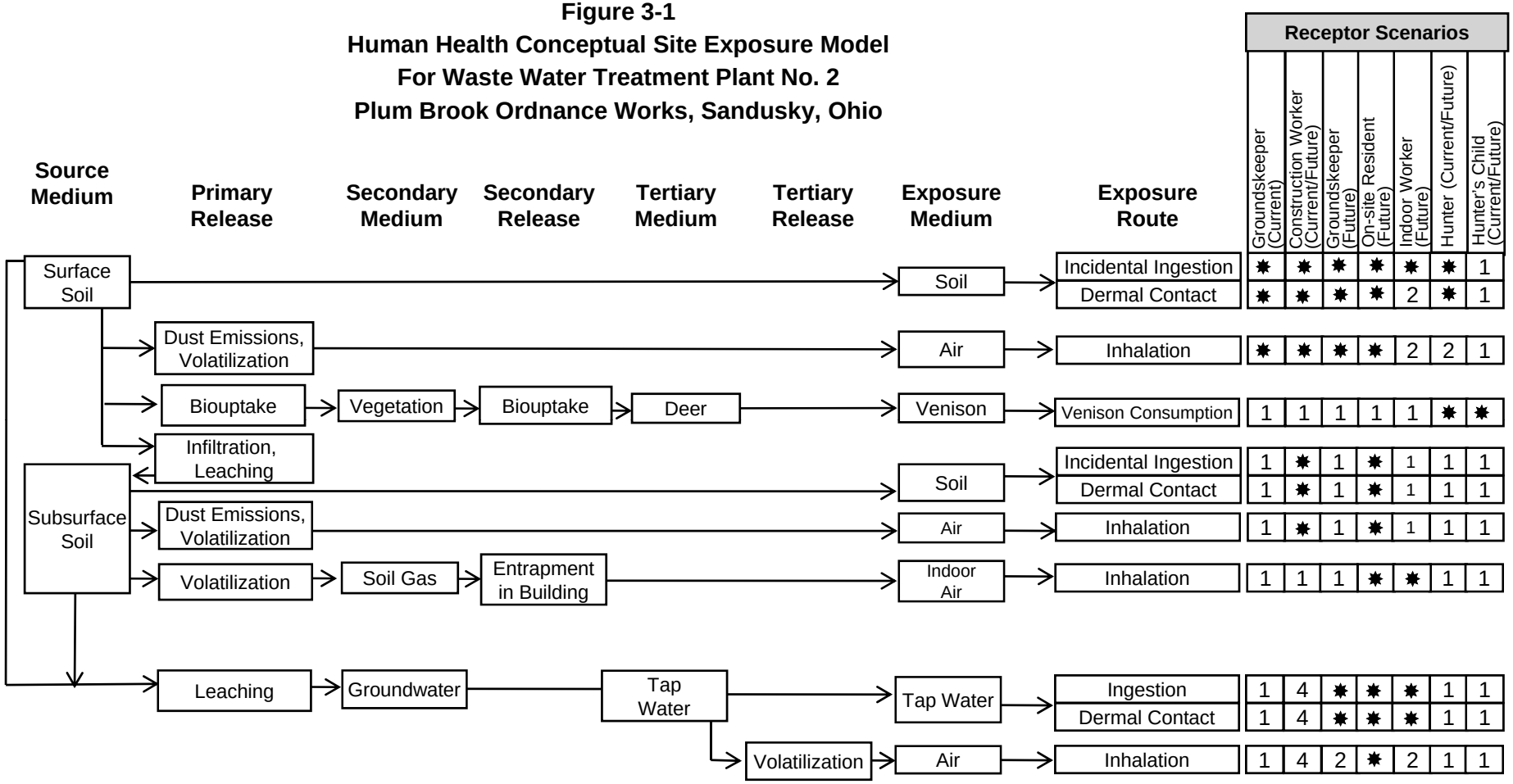
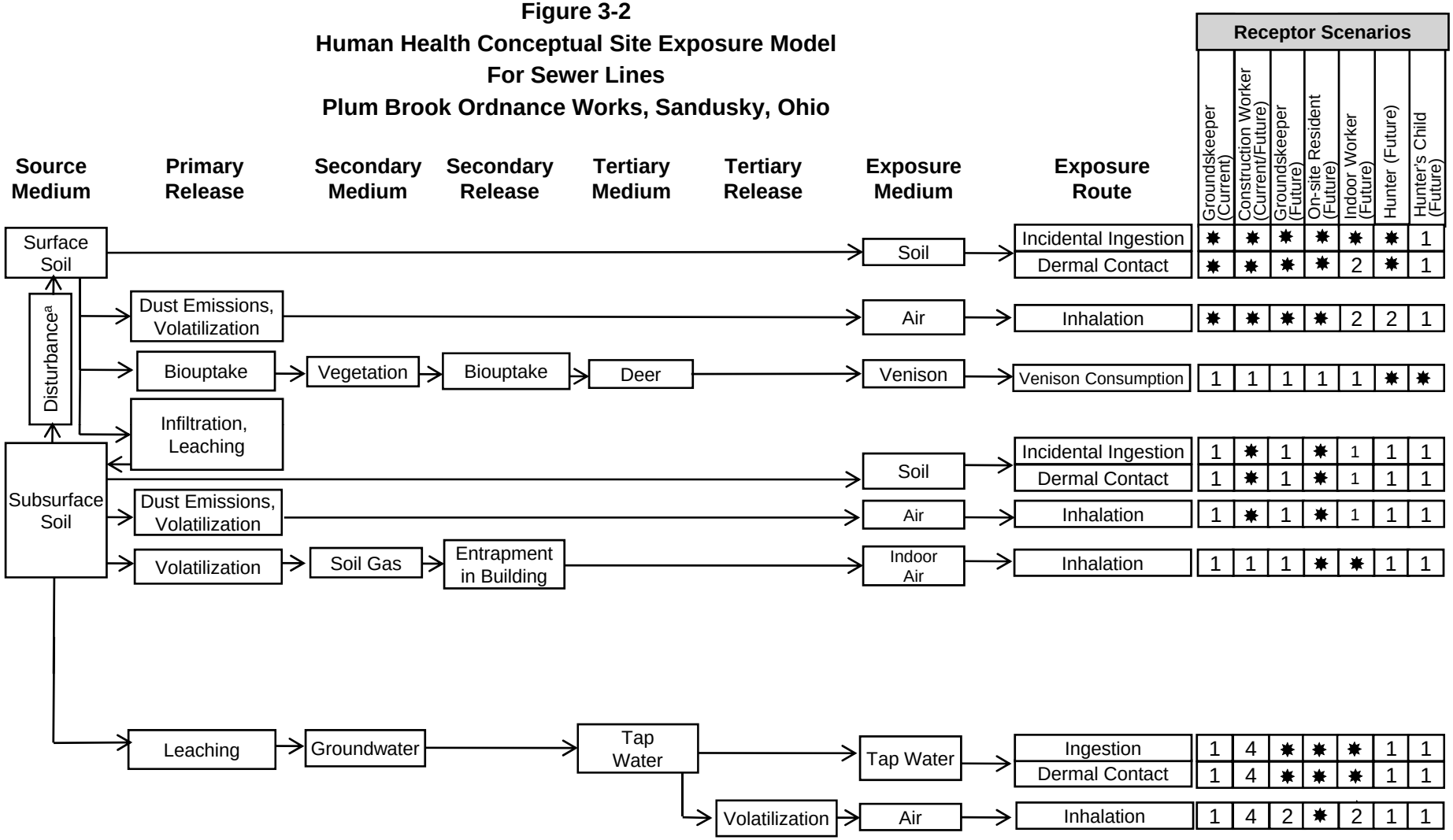
wwtp2_tntcsl_ssl_blhhra_wp_002.dgn cbentley 7/17/2012 2:12:56 PM pdf_with_levels.plt SEL_TEXTSUB_ONLY.TBL

Figure 3-1
Human Health Conceptual Site Exposure Model
For Waste Water Treatment Plant No. 2
Plum Brook Ordnance Works, Sandusky, Ohio



✸ = 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.

Figure 3-2
Human Health Conceptual Site Exposure Model
For Sewer Lines
Plum Brook Ordnance Works, Sandusky, Ohio



[a] Current surface soil may have been historically affected by subsurface soil during sewer line removal.
✸ = 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.

Final
Screening-Level Ecological Risk Assessment
Work Plan
Waste Water Treatment Plant No. 2, TNT Area C to Waste Water Treatment Plant No. 2 Sewer Line, and Waste Water Treatment Plants Steel Sewer Line
Plum Brook Ordnance Works,

Sandusky, Ohio

FUDS Project No. G05OH001818

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June 2013

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List of Acronyms

ARP	assessment receptor profile
BAF	bioaccumulation factor
bgs	below ground surface
BSC	background screening concentration
BTAG	Biological Technical Assistance Group
COPEC	chemical of potential ecological concern
DERP-FUDS	Defense Environmental Restoration Program -Formerly Used Defense Sites
EPA	U.S. Environmental Protection Agency
EPC	exposure point concentration
ESCM	ecological site conceptual model
ESV	ecological screening value
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
PAH	polycyclic aromatic hydrocarbon
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
TNTC	TNT Area C
TNTC/WWTP2 SL	TNT Area C/Waste Water Treatment Plant No. 2 sewer line
TRV	toxicity reference value
UCL	upper confidence limit
USACE	U.S. Army Corps of Engineers
WRS	Wilcoxon Rank Sum
WWTP	waste water treatment plant

List of Acronyms *(Continued)*

WWTP1	Waste Water Treatment Plant No. 1
WWTP2	Waste Water Treatment Plant No. 2
WWTP3	Waste Water Treatment Plant No. 3

1.0 Introduction

This screening-level ecological risk assessment (SLERA) work plan presents the protocol for evaluating the potential for adverse effects posed to ecological receptors from potential hazardous releases from three areas: Waste Water Treatment Plant No. 2 (WWTP2), the trace of the waste water sewer lines which extended from the former TNT Area C (TNTC) to the former WWTP2 (the TNTC/WWTP2 SL), and the trace of the steel sewer line that originated at WWTP2 and connected the other two former waste water treatment plants (WWTP) (the WWTPs Steel Sewer Line). These three areas are 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-Formerly Used Defense Sites (DERP-FUDS). Investigations at PBOW under DERP-FUDS are being managed by the USACE Huntington District and technically overseen by the USACE Nashville District. Although this work plan applies to all three sites, WWTP2, the TNTC/WWTP2 SL, and the WWTPs Steel Sewer Line will be evaluated in separate SLERAs.

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 for TNT Area A (TNTA) and TNTC (IT Corporation [IT], 2001), Wastewater Treatment Plant No. 1 (WWTP1) and Waste Water Treatment Plant No. 3 (WWTP3), and Ash Pits 1 and 3 (Shaw Environmental, Inc. [Shaw], 2009); and the SLERA work plan for the sewer lines extending from TNTA to WWTP1 and from TNT Area B (TNTB) to WWTP1 (Shaw Environmental, Inc. [Shaw], 2010a).

1.1 Facility Description and Location

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. Hunting is allowed by permit on portions of PBOW during the annual deer hunting season.

1.2 Facility History and Background

The PBOW facility was constructed on property comprising 9,009 acres in early 1941 as a manufacturing plant for 2,4,6-trinitrotoluene (TNT), 2,4-dinitrotoluene, and pentolite (USACE, 1995). Production of explosives at PBOW began in December 1941 and continued until 1945. It is estimated that more than 1 billion pounds of nitroaromatic explosives were manufactured during the 4-year operating period. The three explosive manufacturing areas were designated TNTA, TNTB, and TNTC. Twelve process lines were used in the manufacture of TNT, including four lines at TNTA, three lines at TNTB, and five lines at TNTC.

After plant operations ceased, the manufacturing process lines were decontaminated by the Army in late 1945. During decontamination, structures, equipment, and manufacturing debris were either removed and salvaged or removed and burned. After the property was certified as decontaminated, 3,230 acres of the property were initially transferred to the Ordnance Department, then to the War Assets Administration. In 1949, PBOW was transferred to the General Services Administration. This transfer did not include the Plum Brook Depot area, which consists of 2,800 acres. The Department of the Army reacquired the 3,230 acres in 1954. In 1955, the Army completed further decontamination of the manufacturing process lines. This effort included removal of contaminated surface and subsurface soil around the building and wooden and ceramic waste disposal lines containing TNT. Thousands of pounds of TNT were discovered in catch basins; this TNT was removed and burned at the burning grounds. The Army continued cleanup efforts until 1963.

Two property use agreements were entered into by the Army and the National Advisory Committee of Aeronautics, the predecessor of the National Aeronautics and Space Administration (NASA), in 1956 and 1958, respectively. Accountability and custody for the entire portion of the former PBOW property (6,030 acres) that had been under the accountability and custody of the Department of the Army were transferred to NASA on March 15, 1963. NASA performed further decontamination efforts during 1964. The NASA decontamination process included removing contaminated surface soil above the drain tiles, flumes, etc.; destruction of all buildings by fire; and removal of all soil, debris, sumps, and above-grade portions of concrete foundations. Portions of the concrete foundations located below grade were left buried, and some that had been previously slightly above grade were likewise buried. All materials, including the soil in those areas, were flashed. The area was then rough-graded. The decontamination process was also to have included the burning of nitroaromatic-filled flumes that were excavated (Dames & Moore, Inc., 1997).

NASA has operated and maintained the former PBOW property since 1963, and the facility is currently the NASA Glenn Research Center, Plum Brook Station. NASA operates the property as a space research facility in support of their John H. Glenn Research Center at Lewis Field, Cleveland, Ohio. Most of the aerospace testing facilities built at the facility in the 1960s are currently on standby or inactive status. On April 18, 1978, NASA declared approximately 2,152 acres of PBOW as excess. The Perkins Township Board of Education acquired 46 acres of the excess acreage and uses this area as a bus transportation area. The GSA retains ownership of the remaining excess acreage and currently has a use agreement with the Ohio National Guard for 604 acres of this land. NASA currently controls approximately 6,400 acres. The details of land transactions are listed in the site management plan (USACE, 1995).

1.3 WWTP2, TNTC/WWTP2 SL, and WWTPs Steel Sewer Line Description and History

During production in the 1940s, three WWTPs (i.e., WWTP1, WWTP2, and WWTP3) were used to process production waste water from the three TNT manufacturing areas at PBOW. The manufacturing areas were denoted TNTA, TNTB, and TNTC. The wastes were accumulated in the settling basins of the TNT manufacturing areas. These wastes were transported to the waste water treatment plants via aboveground and below-ground wood-stave sewer lines (USACE, 1995). Use of wood-stave pipes was not uncommon for water and sewage conveyance during the late 1800s until the 1950s.

Figure 1-2 shows the areas of concern. WWTP2 is approximately 1.4 acres in size. The lateral extent of influence was assumed to be 10 feet to either side of the sewer line trace for both the TNTC/WWTP2 SL and the WWTPs Steel Sewer Line, resulting in an approximate site size of 0.7 acre and 4.6 acres, respectively. WWTP1 is located nearest to TNTA and received waste water from the settling basins located at TNTA and from TNTB. WWTP2 received waste water from TNTC and is located northeast of the West Area Red Water Pond, east of Acid Area No. 2, and north of TNTC. WWTP3 is located between WWTP1 and WWTP2 and northwest of TNT Area B. The purpose of the plants was to reduce the volume of waste water until it formed a thick liquor that could be incinerated. The waste water that was received from 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, nitrotoulenes, and dinitrotoluenes (Dames and Moore, Inc., 1996).

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 liquor storage tank. WWTP1 and

WWTP2 also included a caustic storage tank and a raw waste storage tank. Both WWTP1 and WWTP2 received waste water from the TNT manufacturing settling basins and neutralized the slurry through a chemical depuration process. The liquid was thickened by evaporation and 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. The incinerator was located south and west of the storage tanks at WWTP2. Ash from the incinerator was disposed of in nearby ash pits associated with the power plants (USACE, 2000). Although incineration is interpreted to be the main treatment process for the waste water, historical design drawings suggest waste water may have been discharged directly to nearby ponds during periods of high production (e.g., the West Area Red Water Pond and Pentolite Road Red Water Pond). This is based on historical drawings showing the presence of a waste water line leading from the raw waste storage tank to the West Area Red Water Pond. In addition, WWTP3 is believed to have been constructed to handle overflow from WWTP1 and WWTP2 since it did not receive any waste water directly from the TNT manufacturing areas.

The only waste line leading into WWTP3 was the WWTPs Steel Sewer Line, which was a 4-inch-diameter steel sewer line that connected WWTP1 and WWTP2 to WWTP3. The layout of WWTP3 was similar to the other WWTPs except that it did not have caustic storage and raw waste water storage tanks. The lack of the caustic storage and raw waste water storage tanks at WWTP3 and the lack of sewer lines directly connecting WWTP3 with any of the TNT manufacturing areas suggest it was used to treat neutralized waste from WWTP1 and WWTP2. The WWTPs Steel Sewer Line was connected to the evaporator buildings at all three WWTPs and is believed to have been used to transfer neutralized waste to WWTP3 during periods of high productivity. Under a previous delivery order, a geophysical survey was conducted at approximately 10 locations along the sewer line. The purpose of the geophysical survey was to determine if the steel sewer line was still present. Results of the geophysical survey indicate the Steel Sewer Line is present and is located at an estimated depth of 3 to 5 feet below ground surface (bgs). Subsequent to the geophysical investigation, actual trenching confirmed the steel sewer line was still present.

During PBOW operations, the sewer lines reportedly often became clogged with TNT residue, and in some instances were completely plugged. The plugged lines were abandoned, and larger diameter bypass sewer lines were constructed around the blocked areas to provide continual drainage of the waste water (USACE, 1995). It is unknown whether clogging was an issue or if bypass lines were constructed for either of the sewer lines (the TNTC/WWTP2 SL or the WWTPs Steel Sewer Line) that are addressed by this work plan.

1.4 Scope and Objectives

SLERAs will be performed to provide an estimate of the potential for adverse ecological effects associated with suspected releases at WWTP2, the TNTC/WWTP2 SL, and the WWTPs Steel Sewer Line. It is noted that SLERAs will be developed for each of these three sites as separate documents; however, the term "SLERA" (singular) is used in this work plan to refer to practices, components, and functions common to all three assessments.

The results of the SLERA will contribute to the overall characterization of the sites 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), *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 releases. This objective will be met by characterizing the ecological communities in the vicinity of the sewer line traces, determining the particular contaminants 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, threatened and endangered species, and other sensitive habitats associated with WWTP2, the TNTC/WWTP2 SL, and the WWTPs Steel Sewer Line. Although water bodies are present at some of the sites being evaluated, they have been addressed as part of other areas of concern at PBOW. Therefore, aquatic receptors will not be evaluated in the SLERA.

Concentrations of chemicals measured in relevant environmental media will be used to perform a SLERA, which will include a 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 of the work plan.

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 site 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 site is minimal or nonexistent due to the lack of potential ecological receptors.
- Potential for adverse ecological effects to ecological entities at the site is minimal or nonexistent due to the lack of potential exposure pathways.
- Potential for adverse ecological effects to ecological entities at the site 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 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, a terrestrial ecological conceptual site model 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

WWTP2 and the TNTC/WWTP2 SL and WWTPs Steel Sewer Line traces and adjacent area will be described in sufficient detail to ensure that technical specialists that review the SLERA can be oriented to the area being investigated. 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 both a spring and fall site reconnaissance for the purpose of collecting qualitative information on the type, quality, and location of biological resources in the vicinity of WWTP2, the TNTC/WWTP2 SL, and the WWTPs Steel Sewer Line. 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 and reptile identifications will be made by sightings.
- The area will be examined for vegetative stress. Stress may be exhibited by stunted growth, poor foliage growth, tissue discoloration, and a loss of leaf coverage.

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 site being evaluated, reconnaissance personnel will obtain relevant information on the vicinity of the historical site features and sewer line traces, 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 traces. Additionally, the *Biological Inventory of Plum Brook Station, 1994* (Ohio Department of Natural Resources, 1995) and the *Protected Species Management Strategy for NASA Glenn Research Center at Lewis Field and Plum Brook Station* (NASA, 2002), which identify and show the locations of threatened and endangered species at PBOW and provide results of extensive wildlife surveys at the facility, 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 area in the vicinity of WWTP2 or the sewer line traces and the probable pathways by which contaminants may be released from the traces to the surrounding environment will be identified. Reconnaissance personnel will use the site visit to evaluate WWTP2 and the sewer line traces for more subtle clues of potential effects from contaminant release.

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 in the vicinity of the traces 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 terrestrial resources at WWTP2 and along the sewer line traces 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.

2.3 Documentation of Potential Receptors of Special Concern and Critical Habitat

A determination will be made as to whether WWTP2, the sewer line traces, and adjacent areas 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. NASA is currently developing a wetland database for the installation, and this database will be consulted as well.

2.4 Significant Ecological Threats

A determination will be made as to whether significant ecological threats exist and whether those threats are related to chemical contamination caused by U.S. Department of Defense activities. The initial screening for significant threats 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 detected site-related chemicals to be carried through the risk assessment. Separate lists of COPECs will be identified for the WWTP2, TNTC/WWTP2 SL, and WWTPs Steel Sewer Line SLERAs. 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

Data will be organized and analyzed separately for the WWTP2, TNTC/WWTP2 SL, and WWTPs Steel Sewer Line. The data for each chemical will be sorted by medium. The TNTC/WWTP2 SL and the WWTPs Steel Sewer Line cross Pipe Creek, and WWTP2 is adjacent to this surface water body (Figure 1-2). Surface water and sediment samples from Pipe Creek have been included and evaluated under other projects (Shaw, 2010b) and will not be repeated in these SLERAs.

For ecological impacts, soil from 0 to 6 feet 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 TNTB and the Red Water Ponds at PBOW (IT, 2000a,b). Chemicals that are not detected at least once in soil will not be included in the risk assessment. Available background data will be determined for soil. 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 4.1 [EPA, 2011]) 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, as such values compromise the ability of ProUCL to accurately determine the distribution of a data set; 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). Please note that the exclusion of high nondetect values that would otherwise result in artificially elevate exposure point concentrations is recommended by EPA (1989).

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 the average of the field duplicate and parent sample if both are positive detections or if both are nondetects.

- Use the 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 this value 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. Generally, chemicals that are detected only at low concentrations in less than 5 percent of the samples from a given medium are eliminated as COPECs unless their presence is expected based on historical information about the site (e.g., nitroaromatics at the TNTC/WWTP2 SL). 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 available for soil at PBOW.

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). It is noted that the method agreed upon for the development of BSCs, as recorded in the September 11, 2002 PBOW Team Meeting minutes, differs from that shown in current OEPA (2009) guidance. This PBOW team background screening method, which has been used for all PBOW risk assessments

over a period of more than a decade, will be used for the WWTP2 and associated sewer line SLERAs to ensure consistency in the evaluation between PBOW sites.. The background data set and derivation of soil BSCs for all PBOW soil investigations are 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 larger number of samples were collected. Chemicals with MDCs less than their respective BSCs 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 soil and ecological screening values (ESV) for ecological endpoints following recommendations received from OEPA and as discussed in EPA *Region 5 Biological Technical Assistance Group (BTAG) 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 along the sewer line traces.

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. Note that separate tables will be provided for WWTP2 and the TNTC/WWTP2 SL and WWTPs Steel Sewer Line traces.

- 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
- BSC
- 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 ESVs, 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 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 separately for WWTP2, the TNTC/WWTP2 SL, and the WWTPs Steel Sewer Line 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. Because nearby water bodies were evaluated in separate investigations (Section 1.4), only terrestrial ecological systems will be evaluated in the SLERA. For terrestrial communities, the major groups are plants and wildlife, including terrestrial invertebrates, mammals, and birds. 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 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 site, (4) sufficient toxicological information available in the literature for comparative and interpretive purposes, (5) sensitivity to COPECs, (6) relatively high likelihood of occurrence on site 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 the species will be present at the site or that habitats present at the site 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 in the vicinity of WWTP2 and along the sewer line traces may 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 appended to the final SLERA report.

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 for each trace 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

represent species that are protected under the Endangered Species Act, species that are candidates for protection, or 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) definability 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 identifying those that are known to be adversely affected by chemicals at the site, 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 site. This information may be included in the ARP for appropriate receptors.

2.9 Ecological Site Conceptual Model

Pictorial representations of the exposure characterization will be prepared for WWTP2 and the TNTC/WWTP2 SL and WWTPs Steel Sewer Line traces in their respective SLERAs. These pictorial representations and any text necessary to clarify the representations will represent the ecological site conceptual models (ESCM). The ESCMs will trace the contaminant pathways through both abiotic components and biotic food web components of the environment. The ESCMs will present all potential exposure pathways and will identify those pathways that are complete and incomplete. The ESCMs will clearly identify the relationship between the measurement and assessment endpoints. They will be used as tools 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.

It should be noted that the WWTP2 (1.4 acres), the TNTC/WWTP2 SL (0.7 acre, assuming 10 feet influence on either side of the sewer line trace), and the WWTPs Steel Sewer Line (4.6 acres, with the same 10-foot area of influence assumption) are relatively small compared with the home ranges of most ecological receptors, which can range from less than an acre to hundreds or even thousands of acres for large-range receptors such as deer. Further, the endpoints of interest for ecological risk assessments typically focus on the protection of local populations of organisms, rather than individuals (unless threatened or endangered species are present, in which case the protection of individual organisms is the appropriate endpoint). Most mobile receptors have home ranges that are similar to, or larger than, the area represented by these sites, and also have territorial or other life history characteristics that limit densities of individuals within a given area. Because of the small sizes of these sites, it is highly unlikely that multiple individuals (i.e., a local population) of a given species would come into regular contact with any potentially contaminated media present at the WWTP2, the TNTC/WWTP2 SL, or the WWTPs Steel Sewer Line. Therefore, even if contamination were present, adverse impacts to populations is not a likely outcome because the potential of exposure to multiple individuals at sites of this size is not sufficient to result in population-level effects. In other words, with the possible exception of those receptors with very small home ranges and relatively high density parameters (e.g., small mammals, invertebrates, plants, etc.), even if site-related contamination were present at these sites, the sites are not large enough to affect populations of most species that may come into contact with the impacted areas. Therefore, it could be argued that performing a SLERA is not necessary, as any potential contamination is likely insignificant from an ecological perspective. However, it is important to note that site-specific SLERAs are performed to satisfy PBOW project administrative requirements, and SLERAs will be performed for WWTP2, the TNTC/WWTP2 SL, and the WWTPs SSL.

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 sewer line traces or WWTP2, considering both current and reasonably plausible future uses of the site, 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 performed in the SLERA 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 soil. Exposure rates for terrestrial wildlife receptors will be based solely upon ingestion of contaminants from this medium 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, dietary composition, etc.) and also use site-specific area use factors, as follows:

$$Total\ Daily\ Dose = \left(\frac{\left(\left[Soil_j * IR_{soil} \right] + \left[\sum_{i=1}^{N} B_{ji} * P_i * IR_{food} \right] \right)}{Body\ Weight} \right) * AUF \qquad \text{Eq. 3.1}$$

where:

$Soil_j$	=	Concentration of COPEC "j" in soil
B_{ji}	=	Concentration of COPEC "j" in food type "i"
IR_{soil}	=	Soil 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.

The first step in estimating exposure rates for terrestrial wildlife involves the calculation of feeding 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 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 incorporated for the receptor species. This information will be summarized and documented in the ARPs. Algorithms will be evaluated for calculating exposure for terrestrial vertebrates that account for exposure via 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 dermal contact of aquatic organisms with contaminated surface water or sediment. However, as stated in Section 2.6.1, surface water and sediment are not evaluated in this SLERA and have been evaluated under other sites at PBOW.

Food web exposure can occur when terrestrial 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.

Selection of appropriate bioaccumulation factors (BAF) is a critical component to food chain modeling. General approaches for BAF selection have been discussed in Sample and Suter (1994), EPA (1999b), U.S. Army Environmental Center (2005), and EPA (2008). An approach that is consistent with these sources will be followed in the selection of BAFs for PBOW. The general hierarchy for selection of BAFs based on types of sources will be as follows:

1. Use of regression equations derived from paired field- or laboratory-based measurements
2. Ratio-derived BAFs developed based on paired data of tissue concentrations compared to media concentrations where the BAF is equal to the tissue concentration divided by the concentration in the abiotic medium

3. Modeled equilibrium partitioning-derived BAFs based on physical or chemical characteristics

4. Assumptions based on values common to chemical class.

Both U.S. Army Environmental Center (2005) and EPA (1999b) support the use of ratio BAFs in preference to equilibrium partitioning-based BAFs, which are typically calculated based on factors such as log octanol-water partition coefficient (K_{ow}) values, fraction of organic carbon in soil, and/or percent of lipids in invertebrates.

Other general recommendations provided in EPA (2008) will also be followed, including the following:

- For selection of ratio-based BAFs, median values are selected over maximum or other high-end BAFs.
- BAFs for accumulation of polycyclic aromatic hydrocarbons (PAH) into mammalian prey are assumed to equal zero due to the high metabolic breakdown of PAHs in mammals.

Regression equations used to calculate prey tissue concentrations of a specific chemical typically take the following general equation form:

$$\text{Ln}\ (C_{food}) = \text{slope value x ln}\ (C_{abiotic_media}) + \text{intercept value} \qquad \text{Eq. 3.2}$$

where:

C_{food}	=	Concentration of chemical in food type
$C_{abiotic_media}$	=	Concentration of chemical in abiotic media.

Ratio BAFs can be generally presented as follows:

$$C_{food} = \text{BAF x}\ (C_{abiotic_media}) \qquad \text{Eq. 3.3}$$

where:

C_{food}	=	Concentration of chemical in food type
$C_{abiotic_media}$	=	Concentration of chemical in abiotic media
BAF	=	Constant.

BAFs calculated based on equilibrium partitioning typically use a physical constant of a chemical to generate a BAF. A generalized form for this calculation would be as follows:

$$\text{Log (BAF)} = \text{slope value} \times \text{Log } (K_{ow}) + \text{intercept value} \qquad \text{Eq. 3.4}$$

where:

Log (BAF) = Log of the BAF for chemical in food type.

BAFs calculated based on equilibrium partitioning will be applied in the same fashion as ratio-based BAFs to generate a tissue concentration value. K_{ow} values needed for BAFs based on equilibrium partitioning will be obtained from EPA's Estimation Program Interface Suite K_{ow}Win software program (available on-line).

Finally, where ratio-based BAFs are missing and where no equilibrium partitioning method has been developed for calculating BAFs, other methods, such as using BAFs for chemicals in the same class as surrogates, may be presented for establishing ratio-based BAFs.

Media-Specific Exposure Pathways. Only exposure pathways associated with soil will be addressed in the SLERA. As discussed previously, surface water and sediment are not addressed in the SLERA. Although groundwater represents a potential transport medium for COPECs, groundwater itself is not an exposure point in ecological risk assessments.

Soil exposure pathways are potentially important for terrestrial plants and animals along the sewer line traces. For nonburrowing 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.

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. Separate intake summaries will be provided in the WWTP2, TNTC/WWTP2 SL, and WWTPs Steel Sewer Line SLERAs. 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 the Los Alamos National Laboratory (2010) *ECORISK Database (Release 2.5)* and (2) lethal dose 50 percent values from databases 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 for developing TRVs 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).

4.2 Development of Toxicity Reference Values

TRVs will be identified in the literature, when available for each chemical selected as a COPEC at the WWTP2 and the sewer line traces. 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 is judged to be an appropriate toxicological endpoint because 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 for a site-associated COPEC are unavailable, 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 developed 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 from 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 Preventive Medicine (2010) and EPA's *Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (EPA, 1999c). Because NOAELs and LOAELs for the selected wildlife receptor species are based on data from test species that are usually different from the species of concern, a mathematical adjustment to the TRVs has often been performed in the past (e.g., Sample, et al. [1996]) using a power function of the ratio of body weights. This practice is often referred to as allometric scaling. Alternately, uncertainty factors have also been used to account for the differences in species' sensitivities to chemicals. However, in recent years, these practices have been discouraged by most scientific and regulatory groups. Recent reviews of these practices (e.g., EPA [2008]; Allard, et al. [2009]) have concluded that the use of allometric scaling of TRVs does not reflect a sound application of toxicological or ecological risk practices because supporting data for this practice are limited, and the ratio relationships used for the mathematical conversions were developed based on acute (rather than chronic) toxicity data. These reviews further conclude that uncertainty factors to account for inter-species differences based on an arbitrary multiplier should not be used without a scientific basis for their application (Allard, et al., 2009). Therefore, the use of toxicity data without adjustments as reported in the literature is regarded as the most technically sound approach and will be the adopted practice for the SLERA.

Exposure rate TRVs provide a reference point for the comparison of toxicological effects upon exposure to a contaminant. To complete this comparison, TRVs are compared to the doses representing the exposure for each receptor to site contaminants.

5.0 Risk Characterization

The risk characterization phase integrates information on exposure, exposure-effects relationships, and defined or presumed target populations. The result is an estimation of the potential for these populations to experience adverse effects resulting from exposure to environmental stressors present at a site. A semiquantitative 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 performed using the comparison of the MDCs 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. Ecological risk will be characterized separately for each of the sites in the WWTP2, TNTC/WWTP2 SL, and WWTPs Steel Sewer Line SLERAs.

5.1 Hazard Estimation

The hazard estimation for WWTP2 and the TNTC/WWTP2 SL and WWTPs Steel Sewer Line traces 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., PAHs, 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 in determining 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. 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 1.0, 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 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 site. 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 on site (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 uncertainties analysis of the SLERA will describe guidelines for interpreting the potential risk posed from contaminants. The following limitations associated with HQs are noted and will be briefly addressed in the final SLERA report (Tannenbaum et al., 2003, and Tannenbaum, 2005):

- 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 in the SLERA: (1) summary of the ecological risks associated with the sites and (2) interpretation of 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 each 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 each 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.

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FIGURES

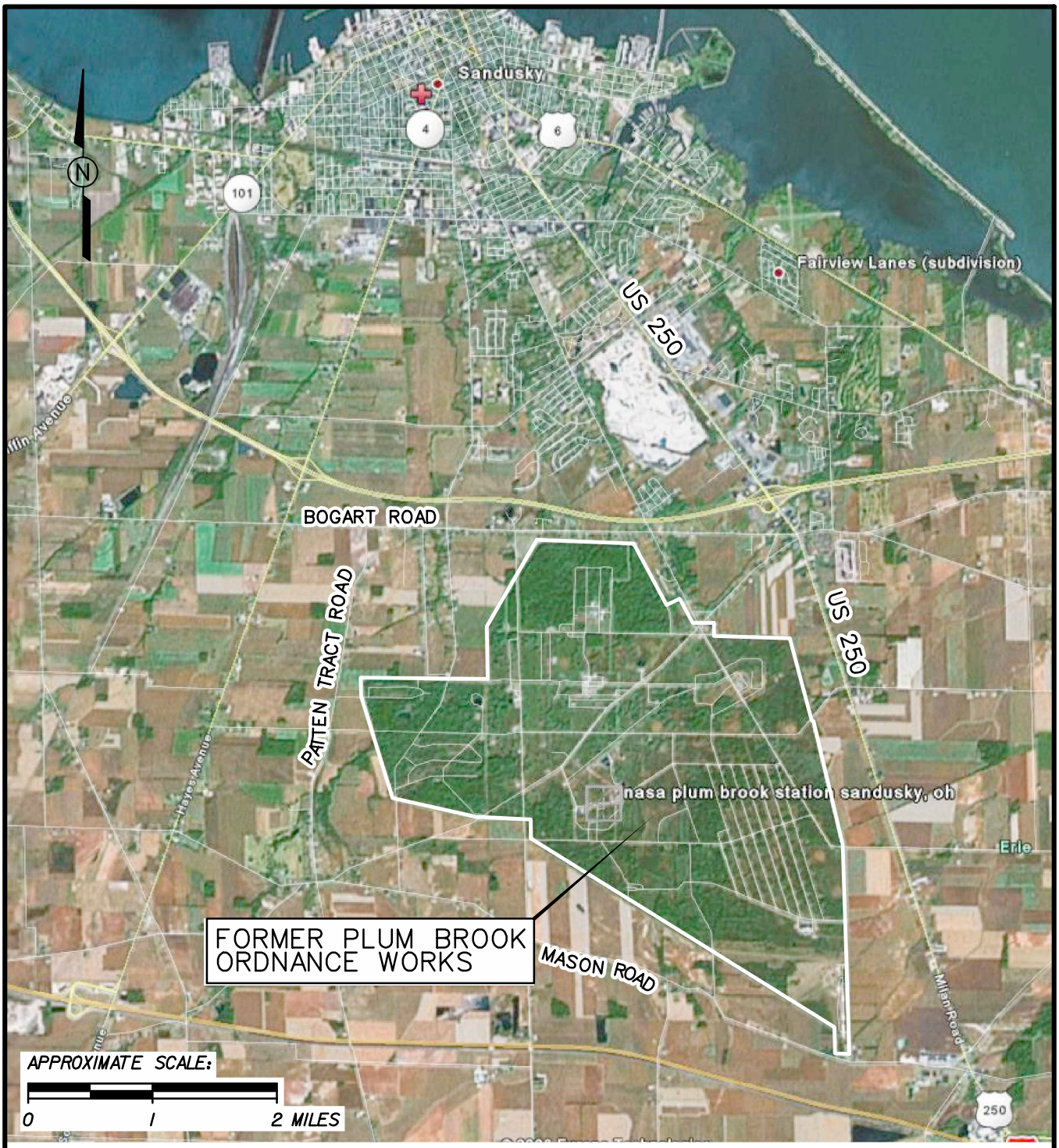


FIGURE 1-1
PBOW VICINITY MAP

WWTP2, TNTC/WWTP2 SL, AND WWTP2 SSL SCREENING-LEVEL ECOLOGICAL RISK ASSESSMENT WORK PLAN
FORMER PLUM BROOK ORDNANCE WORKS
NASA PLUM BROOK STATION
SANDUSKY, OHIO

Shaw® a world of Solutions™

wwtp2_tntcsl_ssl_slerd_wp_001.dgn cbentley 7/12/2012 1:30:50 PM pdf_with_levels.plt SEI_TEXTSUB_ONLY.TBL

NOTES:

1. WWTP2 = WASTE WATER TREATMENT PLANT No.2.
2. TNTC/WWTP2 SL = TNT AREA C TO WASTE WATER TREATMENT PLANT No.2 SEWER LINES.
3. WWTP2 SSL = WASTE WATER TREATMENT PLANT No.2 STEEL SEWER LINES.

LEGEND:

- AREA OF CONCERN
- APPROXIMATE LOCATION OF SEWER LINES
- POND
- CREEK, DITCH, CONVEYANCE
- RAILROAD
- ROAD
- FENCE
- FACILITY BOUNDARY

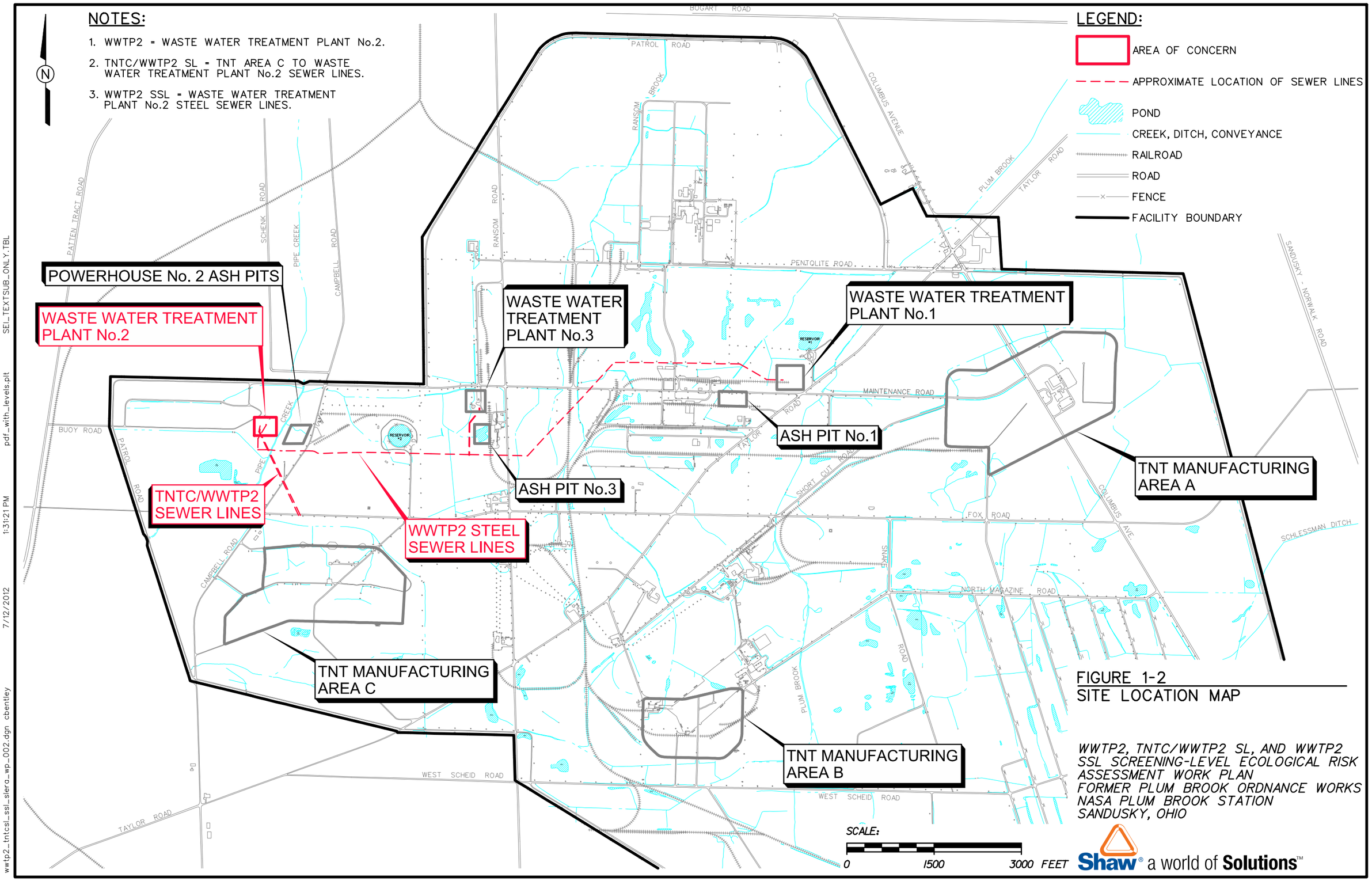


FIGURE 1-2
SITE LOCATION MAP

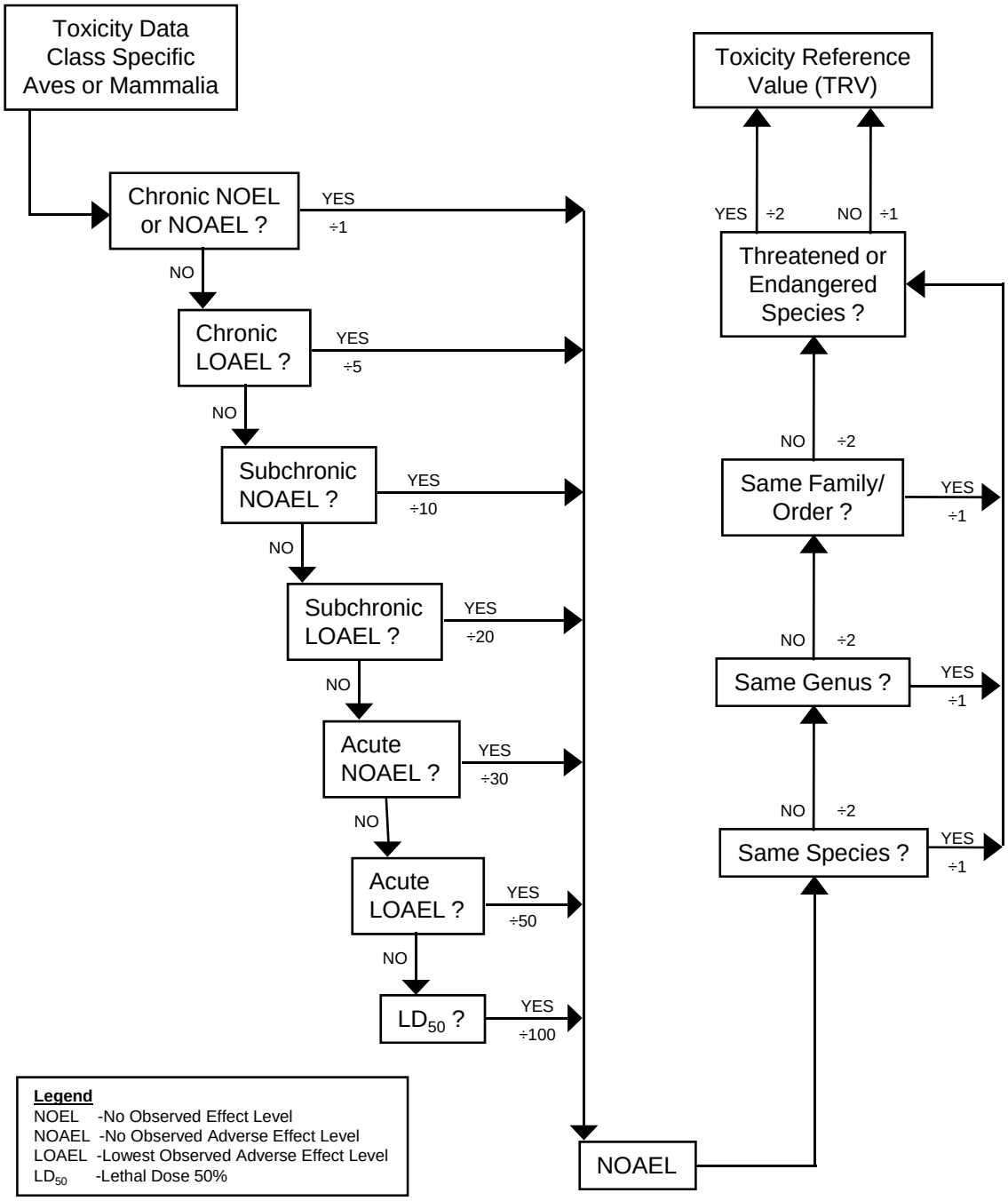
WWTP2, TNTC/WWTP2 SL, AND WWTP2 SSL SCREENING-LEVEL ECOLOGICAL RISK ASSESSMENT WORK PLAN
FORMER PLUM BROOK ORDNANCE WORKS
NASA PLUM BROOK STATION
SANDUSKY, OHIO

Shaw® a world of Solutions™

wwtp2_tntcsl_ssl_slero_wp_002.dgn cbentley 7/12/2012 1:31:21 PM pdf_with_levels.plt SEL_TEXTSUB_ONLY.TBL

Figure 4-1

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



Credit: Adapted from Ford et al. (1992) in *Tri-Service Procedural Guidelines for Ecological Risk Assessments*, 1996

RESPONSE TO COMMENTS

Responses to Ohio Environmental Protection Agency Comments on the Draft Baseline Human Health Risk Assessment and Screening Level Ecological Risk Assessment Waste Water Treatment Plant No. 2, TNT Area C to Waste Water Treatment Plant No. 2 Sewer Lines, and Waste Water Treatment Plant No. 2 Steel Sewer Line Plum Brook Ordnance Works, Sandusky, Ohio, Dated July 18, 2012 FUDS Project No. G05OH001818

Comments from Janusz Byczkowski, Ph.D., Toxicologist*, Division of Environmental Response and Revitalization, Ohio Environmental Protection Agency, received September 21, 2012.*

Note: Comments refer to Sections of the Baseline Human Health Risk Assessment (BHHRA) or the Screening Level Ecological Risk Assessment (SLERA) as noted.

Comment 1: **BHHRA Section 2.4.1, Page 2-4, Line 19. <u>This document states:</u> "*...Because the RSL of 15 µg/L can be used to screen for a potential average concentration of 15 µg/L, the RSL of 15 µg/L is used as the RBSC...*" This statement sounds trivial. Please rephrase.**

Response 1: This text will be revised consistent with the following: "*Because the Office of Drinking Water action level of 15 µg/L can be used to conservatively screen for a potential average concentration of 15 µg/L, this concentration is used as the RBSC.*"

Comment 2: **BHHRA List of Acronyms, Page iv, and Section 3.3.3.3, Page 3-19, Line 7. <u>This document states:</u> "*…à attenuation coefficient...*" (Latin small A with acute) and again in section 3.2.2.2, P. 3-19: " *...(à) ...*" (Latin small A with acute); then on P. 3-21 in Equation 3.6 "*...α = ...*" (alpha) and below: "*...where:*

α = attenuation coefficient..." (alpha). Then again in Equation 3.13 "$C_{building} = \alpha$ ..." (alpha). Please correct typos and use uniform symbols for attenuation coefficient throughout the document.**

Response 2: This apparent software conversion error will be corrected so that the parameter is "*α*."

Comment 3: **BHHRA Section 3. 3.5, Page 3-34, Equation 3.32. <u>This document states:</u> Below the equation 3.32 "*...where: IRv = ingested dose of COPC...* [...] *IRv = venison ingestion rate...*" The same variable cannot be defined twice in two different ways. Please correct definition of variables.**

Response 3: The first "IR_v" should be "I_v". This will be corrected in the final report.

Comment 4: **BHHRA Section 9.0, Page 9-1, Line 4; Page 9-5, Line 4, and Page 9-5, Line 7, and Table 3-2, Page 4, Line 19. <u>This document states:</u>**

"… http://www.oehha.co.gov/risk/ChemicalDB/index.asp …"
Also
"… http://www.epa.gov/superfund/programs/lead/products.htm …"
and
"… http://www.epa.gov/region09/waste/sfund/prg/files/04usersguide.pdf …"

Some Internet links do not work. Please check and correct/update URLs.

Response 4: The URLs will be updated. The correct URLs are as follows:
http://oehha.ca.gov/risk/chemicaldb/index.asp
http://www.epa.gov/superfund/lead/pbrisk.htm
http://www.epa.gov/region9/superfund/prg/files/04usersguide.pdf.

Comment 5: **SLERA, Section 2.6.2, Page 2-5, Line 13. Page 4-1, Section 4.2. <u>This document states:</u> "…*Nondetects with method detection limit greater than the MDCs will not be included in the data set used to calculate the EPC (EPA, 1989)*…" This statement is not clear and seems to do not reflect the quoted reference (EPA, 1989). If this is related to the case discussed in RAGS Vol. I (EPA, 1989), Chapter 5, Sect. 5.3.2 (P.5-10), it should be rephrased accordingly. Please rephrase in accordance with the reference quoted (EPA, 1989).**

Response 5: EPA (1989) was cited because it allows that samples with elevated nondetect values may be excluded from the quantitative risk assessment. This is particularly important in the use of ProUCL where some (but not all) of the statistical methods can assume only one detection limit. In these methods, ProUCL recognizes only the largest detection limit entered for the data set and then assumes that all observations (whether detected or nondetect) less than the highest detection limit are nondetects. Thus, the inclusion of elevated detection limits can artificially limit the power of these methods and can limit the overall usefulness of ProUCL.

Therefore, to clarify, the text consistent with the following sentence will be added at the end of the first paragraph of Section 2.6.2: "*Please note that the exclusion of high nondetect values that would otherwise result in artificially elevated exposure point concentrations is recommended by EPA (1989)*." Also, the reference to "(EPA, 1989)" will be removed from sentence that is referenced by the reviewer. As stated above, inclusion of an elevated detection limit under ProUCL may not only artificially inflate the upper confidence limit estimate, but may also unduly limit the performance of ProUCL. If such values were to be excluded from the SLERA, these would be clearly identified and discussed as an uncertainty.

Comment 6: **BHHRA Section 2.4.3.1, Page 2-6, Line 21, and SLERA Section 2.6.4, Page 2-6, Line 28. <u>This document states:</u> "…*This PBOW Team agreement, which has been used for all PBOW risk assessments to date, takes***

precedence over the subsequent OEPA (2009c) guidance…" **Please provide justification for not applying in BHHRA and SLERA the recommended OEPA-DERR (2009) methodology.**

Response 6: The fifth sentence in the first paragraph of Section 2.4.3.1 (beginning "This PBOW team agreement…") will be replaced with text that is consistent with the following: "*This PBOW team background screening method, which has been used for all PBOW risk assessments over a period of more than a decade, will be used for the WWTP2 and associated sewer line BHHRAs to ensure consistency in the evaluation between PBOW sites."*