Baseline Human Health Risk Assessment Work Plan and Ecological Risk Assessment Work Plan Red Water Ponds Areas Plum Brook Ordnance Works Sandusky, Ohio

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List of Acronyms

ABS	absorption factor
AF	adherence factor
ARAR	applicable or relevant and appropriate requirement
AT	averaging time
ATSDR	Agency for Toxic Substances and Disease Registry
BW	body weight
CDI	chronic daily intake
COC	chemical(s) of concern
COPC	chemical(s) of potential concern
CSEM	conceptual site exposure model
CT	central tendency
DA	dose absorbed
DAF	dilution attenuation factor
DNT	dinitrotoluene
DOD	U.S. Department of Defense
DOE	U.S. Department of Energy
ED	exposure duration
EF	exposure frequency
EPA	U.S. Environmental Protection Agency
ET	exposure time
FI	fractional term
ft bgs	feet below ground surface
g	gram
GAF	gastrointestinal absorption factor
HEAST	Health Effects Summary Assessment Table
HI	hazard index
HQ	hazard quotient
ILCR	incremental lifetime cancer risk
IRIS	Integrated Risk Information System
IT	IT Corporation
kg	kilogram
K_{ow}	octanol-water partition coefficient

List of Acronyms *(Continued)*

L	liter
m^3	cubic meter
MDC	maximum detected concentration
µg	microgram
mg/cm^2	milligrams per square centimeter
mg/day	milligrams per day
NASA	National Aeronautics and Space Administration
NCEA	National Center for Environmental Assessment
OEPA	Ohio Environmental Protection Agency
PAH	polynuclear aromatic hydrocarbon
PBOW	Plum Brook Ordnance Works
PC	permeability coefficient
PCDD/PCDF	polychlorinated dibenzo-p-dioxin/dibenzofuran
PEF	particulate emission factor
PRG	preliminary remediation goal
QC	quality control
RA	risk assessment
RBRC	risk-based remediation criteria
RBSC	risk-based screening concentration
RfC	reference concentration
RfD	reference dose
RME	reasonable maximum exposure
SA	surface area
SF	slope factor
SQL	sample quantitation limit
SSL	soil screening level
SVOC	semivolatile organic compound
TNT	trinitrotoluene
UCL	upper confidence limit
URF	unit risk factor
USACE	U.S. Army Corps of Engineers
UTL	upper tolerance limit
VOC	volatile organic compound

1.0 Introduction

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

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 (USACE, 1997). PBOW is located within the transition of the two systems. Upwards of 170 private drinking water wells permitted by the Erie County Health Department are located within 4 miles of PBOW. Permits are not required for agricultural wells. The Erie County Health Department does not permit using surface water as private drinking water. Lake Erie and Sandusky Bay are used for recreational swimming, fishing, and boating.

Current land uses of the PBOW facility are classified as industrial for the purpose of identifying plausible human receptors and exposure pathways for evaluation in the risk assessment (RA). USACE describes potential future uses of all or portions of the facility as:

- Continued industrial use (NASA activities and programs)
- Recreational use of portions of the site by hunters and fishermen

- Sale of portions of the site to state or local government or private individuals (no land-use restrictions were mentioned)
- Possible use of parts of the facility for residential or agricultural purposes
- Possible use of parts of the facility for training by the National Guard.

Construction activities may be performed during development of any of the sites.

In summary, future land use of the Red Water Ponds Areas is considered to be industrial or residential for the purposes of this evaluation. It is assumed that bedrock groundwater may be developed as a source of potable water. The shallow or perched groundwater is transient and not considered a potential source of potable water. There are no plausible pathways by which receptors would be exposed to this medium.

Bedrock groundwater is a medium of interest for the Red Water Ponds Area, and exposure thereto is included in this work plan, although it is understood that analytical data will not be available for evaluation when the RA is performed. The RA, therefore, will be incomplete, because total risks and hazards across media cannot be determined. The protocol for groundwater evaluation is included, however, to underscore the need for its eventual evaluation to complete the RA, and to obtain regulator approval of the protocol to expedite completing the risk assessment when data become available at a later date.

The purpose of this work plan is to describe the protocol for evaluating risk to human health at the two Red Water Ponds Areas at PBOW: the West Area Red Water Ponds and the Pentolite Road Red Water Pond Area. Earlier investigations summarized by USACE indicate that subsurface soil, surface water, the shallow or overburden groundwater and the bedrock groundwater aquifer are contaminated with organic compounds including explosives (predominantly TNT degradation products), volatile organic compounds (VOC), semivolatile organic compounds (SVOC) and possibly metals.

This work plan is intended to serve as the template for the RA report. A RA is a stand-alone document, chapter or section; i.e., all the equations and values necessary for quality control (QC) and replication of computations must be contained within the report itself.

The risk assessment work plan is based on U.S. Environmental Protection Agency (EPA), USACE, and Ohio Environmental Protection Agency (OEPA) guidance, including, but not limited to, the following:

- Ohio Environmental Protection Agency (OEPA), 1993, *Closure Plan Review Guidance for RCRA Facilities*, Interim Final, OEPA Division of Hazardous Waste Management, September 1.

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

- 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: 9285.6-03.

- EPA, 1992a, *Supplemental Guidance to RAGS: Calculating the Concentration Term*, Office of Solid Waste and Emergency Response, Washington, DC, Publication 9285.7-081.

- EPA, 1992b, *Dermal Exposure Assessment: Principles and Applications*, Interim Report, Office of Research and Development, Washington, DC, EPA/600/8-91/011B, including Supplemental Guidance dated August 18, 1992.

- EPA, 1992c, "Guidance on Risk Characterization for Risk Managers and Risk Assessors," Memorandum from F. Henry Habicht II, Deputy Administrator, to Assistant Administrators, Regional Administrators, February 26, 1992.

- USACE, 1995, *Risk Assessment Handbook, Volume I: Human Health Evaluation*, Engineer Manual EM 200-1-4.

The remainder of this document is organized as follows. Chapter 2.0, Data Evaluation, describes the selection of chemicals of potential concern (COPC) for each medium of interest, and estimation of source-term concentrations for each COPC in each medium. Chapter 3.0, Exposure Assessment, describes the exposure scenarios and the rationale by which plausible receptors are selected, the pathways by which they may be exposed, the exposure-point concentrations of COPC, and the estimated dose or contact rates for each of the COPC. Chapter 4.0, Toxicity Evaluation, describes the hazard evaluation, i.e., the adverse health effects associated with each of the COPC, and the dose-response evaluation, i.e., the relationship between dose or contact rate and the magnitude of the adverse effect. Chapter 5.0, Risk Characterization, combines the output

of the exposure analysis and the toxicity analysis to quantify cancer risk and noncancer hazard to each receptor, identifies chemicals of concern (COC), which are the chemicals responsible for unacceptable risk or hazard estimates, develops applicable or relevant and appropriate requirements (ARAR) for the COC, and develops risk-based remediation criteria (RBRC) for the COC. Chapter 6.0, Uncertainty Analysis, describes the uncertainty associated with the components of the RA, and presents the Monte Carlo simulation to illustrate the uncertainty about the point risk and hazard estimates. Chapter 7.0, References, presents the references used in the preparation of this document.

2.0 Data Evaluation

2.1 Identification of COPC

Prior to initiation of a baseline RA, a list of chemicals present in site samples will be compiled. This initial list includes all chemicals detected in any site medium. From this list, COPC are selected as follows:

2.1.1 Sorting the Analytical Data

The chemicals evaluated in a given medium are limited to those detected in that medium. Chemicals for which all samples yield nondetects are considered not to be present and are not evaluated further. The data for each chemical will be sorted by medium. Surface soil (0 to 1 feet below ground surface [ft bgs]) and subsurface soil (1 to 10 ft bgs for direct exposure pathways) will be considered separate media.

2.1.2 Evaluating Data Quality

The analytical data may have qualifiers from the analytical laboratory QC 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 (EPA, 1989a):

- 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 - QC indicates that the data are unusable (chemical may or may not be present).
- B - Inorganic chemicals: the concentration is less than the contract-required detection 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 five-times, ten-times (5x, 10x) rule. A chemical is considered a nondetect unless its concentration exceeds five times the blank concentration. For common laboratory contaminants (acetone, 2-butanone [methyl ethyl ketone], methylene chloride, toluene and the phthalate esters), the sample concentration must exceed ten times the blank concentration.

"J" qualified data are used in the RA; "R" and "B" qualified data are not. The handling of "U" qualified data (nondetects) in the RA is described in Section 2.2. 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 RA. If confidence is high that the chemical is present, but the actual concentration is somewhat in question, the data generally are used in the RA.

2.1.3 Identifying Site-Related Chemicals

Identifying site-related chemicals is a matter of professional judgement that must be addressed for each chemical individually. For most organic chemicals, their identification at concentrations above levels in blanks (considering the 5x, 10x rule; see Section 2.1.2) is presumptive evidence of site-related activity. However, there are exceptions. Inorganic or organic chemicals that are detected infrequently may be artifacts in the data that do not reflect site-related activity and should not be included in the RA. Generally, chemicals that are detected only at low concentrations in less than 5 percent of the samples from a given medium are dropped from further consideration, unless their presence is expected based on historical information about the site. Chemicals detected infrequently at high concentrations may identify the existence of "hot spots" and are retained in the evaluation, unless other information exists to suggest that their presence is unlikely to be related to site activities.

Other exceptions among the organic chemicals include background levels of pesticides and herbicides in an agricultural area in which these chemicals are or were used in crop production. Polynuclear aromatic hydrocarbons (PAH), a class of organic compounds which form from natural or anthropogenic combustion of organic matter, including fossil fuels, may be another exception. PAHs generally are ubiquitous in the environment, and background levels in urban, rural, and agricultural soil have been compiled (Agency for Toxic Substances and Disease Registry [ATSDR], 1993).

In addition, any class of organic compound may be considered to be of anthropogenic background if site concentrations are comparable to upgradient concentrations. For example, if concentrations of 2,4-DNT in the water in a creek meandering across (hypothetical) Site A are comparable to upgradient concentrations, it is probably appropriate to conclude that 2,4-DNT is not a site-related chemical. Although the chemical is clearly related to former PBOW activities, its presence at Site A probably does not reflect the activities that took place at that particular site. Since the 2,4-DNT in surface water does not reflect activities at Site A, it is inappropriate to select the compound as a COC for Site A and to develop RBRC for it, because remediation at Site A will not address the source of the contamination. Organic chemicals identified as anthropogenic background will not be eliminated from the RA; instead, they are included and

evaluated in Total Site Risk and Background Risk, but not in Site-Related Risk. These designations are described fully in Chapter 5.0, Risk Characterization.

Resolving the site-related issue for metals is more difficult, because metals are naturally present in most environmental media. Historical data regarding site activities, processes, disposal practices and inadvertent releases can provide much useful information, particularly to confirm the designation of a metal as site related. Eliminating a metal from the site-related list, however, requires confidence in the adequacy of the historical data. Frequently the historical data are incomplete; therefore, statistical techniques are often used as tools to aid the exercise of professional judgement. The statistical techniques generally involve comparing the site data with background data.

The first statistical technique is the development of an upper tolerance limit (UTL) for background, and comparing the maximum detected concentrations (MDC) with the UTL. The UTL is the concentration, with a probability of 0.95 (or a confidence of 95 percent), that will capture (or cover) 95 percent of background samples if a large number of samples were taken. If the site data truly reflect background, there is a 5 percent probability that any site sample concentration will exceed the UTL. Chemicals with MDCs less than the background UTL are eliminated from further consideration. If the MDC exceeds the UTL, the chemical is retained as a COPC, or a more rigorous statistical analysis may be performed. The statistical analysis consists of comparing the site and background data sets to determine if both are drawn from the same population. The Wilcoxon Rank Sum test (STATISTICA™) (EPA, 1992d) is used for this purpose, because it is applicable to all data distributions.

The UTL is calculated as follows:

$$UTL = \overline{X} + k(a) \qquad \text{Eq. 2.1}$$

where:

UTL = upper tolerance limit (confidence factor of 0.95 and coverage of 95 percent)
$\bar{x}$ = arithmetic mean
a = standard deviation

k = tolerance factor (Table 5, OEPA [1991], Final *How Clean is Clean Policy*, 26 July.

It must be understood that statistical analysis is only a tool to aid the exercise of professional judgement. Site data from uncontaminated areas with concentrations at the high end of background may "fail" statistical testing because of the limitations of sample size; i.e., the full range of actual background and site variation was not captured. Statistical testing is based on absolute values, but the approximately 20 metals generally analyzed together constitute only about 4 to 5 percent of a given sample. High values of one or more metals may arise from a diminished amount of other constituents in soil, e.g., silica or organic matter, that may be more abundant in background areas. Therefore, it may be necessary to express the concentration of each metal in each sample as a percent of the total metal content of the sample before performing the comparisons.

2.1.4 Risk-Based Screening

Risk-based screening for human health is introduced to focus the assessment on the chemicals that may contribute significantly to overall risk. In this screen, chemical concentrations are compared with very conservative levels derived for standard exposure scenarios. The MDC is compared with the appropriate risk-based screening concentration (RBSC). If the MDC is less than or equal to the RBSC, the chemical in this medium is not considered further. If the MDC exceeds the RBSC, the chemical is considered to be a COPC

Chemicals whose concentrations are below the RBSC are not considered further in the RA because it is very unlikely that they would cause significant risk. RBSCs for soil, sediment, and groundwater are EPA (1998a) Region IX preliminary remediation goals (PRG) adjusted to reflect an incremental lifetime cancer risk (ILCR) of 1E-7 and a hazard index (HI) of 0.1. Soil and sediment contaminant concentrations are compared with "residential soil" RBSCs, and groundwater contaminant concentrations are compared with "tap water" RBSCs. Surface water concentrations are compared with the tap water RBSCs previously described, if plausible exposure pathways are limited to direct contact (incidental ingestion and dermal contact) in a casual contact, wading, or swimming scenario. It is assumed that total uptake by these pathways would not exceed that from ingestion of 2 liters (L) of water per day, the basis for the EPA PRGs.

The risk-based screening previously described assumes that the RBSCs reflect a sufficiently conservative evaluation of the relevant exposure pathways. The soil RBSCs, however, may not be sufficiently conservative to screen sediment in water bodies from which fish are harvested and consumed, because they do not address the indirect pathway (bioaccumulation by fish). Similarly, the tap water RBSCs may not be sufficiently conservative to screen surface water from which fish are taken. Therefore, mercury and those organic chemicals known to bioaccumulate in aquatic food chains (i.e., organochlorine pesticides, polychlorinated biphenyls, and polychlorinated dibenzo-p-dioxins/dibenzofurans [PCDD/PCDF]) will be selected as COPC in sediment and surface water in surface water bodies from which fish are taken, even if their MDCs are below their RBSCs. Fish consumption is not a concern at the Red Water Ponds Areas because the surface water bodies do not support sport or subsistence fishing.

2.1.5 Evaluating Essential Nutrients

Essential nutrients such as calcium, chloride, iodine, iron, magnesium, phosphorus, potassium, and sodium may be eliminated as COPC, provided that their presence in a particular medium is judged to be unlikely to cause adverse effects on human health.

2.2 Developing Source-Term Concentrations

2.2.1 Soil, Surface Water, and Sediment

Because of the uncertainty associated with characterizing contamination in environmental media, both the mean and the UCL on the mean are usually estimated for each COPC in each medium of interest. The upper 95 percent confidence limit on the mean is generally referred to as the UCL. In general, unusually high values are included in the calculation of the UCL because high values seldom appear as statistical outliers in environmental data. Inclusion of outliers increases the overall conservatism of the risk estimate, or may identify areas that require evaluation as hot spots.

Data sets consisting of five or more data points are tested for normality and lognormality with the Shapiro-Wilks test (EPA, 1992d) using the software package STATISTICA™. Statistical analysis is performed only on those chemicals whose MDCs exceed their RBSCs. Either a normal or lognormal UCL is calculated, whichever provides the better fit in the Shapiro-Wilks test. A nonparametric confidence limit is used when the data fit neither a normal or lognormal distribution.

The UCL is calculated for a normal distribution as follows (EPA, 1992a):

$$UCL = \bar{x} + t_{1-\alpha,n-1} x (s/\sqrt{n}) \qquad \text{Eq. 2.2}$$

where:

UCL = upper 95th confidence limit on the arithmetic mean concentration (calculated)
$\bar{x}$ = sample arithmetic mean
t_1 = critical value for Student's *t*-test
α = 0.05 (95 percent confidence limit for a one-tailed test)
n = number of samples in the data set
s = sample standard deviation.

The UCL is calculated for a lognormal distribution as follows (Gilbert, 1987):

$$UCL = e^{\left(\bar{y} + (0.5 \cdot s_y^2) + \left[H_{0.95} \cdot \frac{s_y}{(n-1)^{0.5}}\right]\right)} \qquad \text{Eq. 2.3}$$

where:

UCL = upper 95th confidence limit on the arithmetic mean concentration (calculated)
$\bar{y}$ = $\sum y/n$ = sample arithmetic mean of the log-transformed data, y = ln x
s_y = sample standard deviation of the log-transformed data
n = number of samples in the data set
$H_{0.95}$ = value for computing the one-sided upper 95 percent confidence limit on a lognormal mean from standard statistical tables (Land, 1975).

The data point selected as the nonparametric UCL is estimated as the 95 percent UCL rank order on the arithmetic mean of the data set. It is estimated by ranking the data observations from smallest to largest. The arithmetic mean is converted to a percentile by interpolation. The rank order of the data point selected as the UCL is estimated from the following equation (Gilbert, 1987):

$$u = p(n+1) + Z_{1-\alpha} \sqrt{np(1-p)} \qquad \text{Eq. 2-4}$$

where:

u = rank order of value selected as upper confidence limit, calculated

p	=	percentile corresponding to the arithmetic mean
n	=	number of samples in the data set
α	=	confidence limit (95 percent)
$Z_{1-\alpha}$	=	normal deviate variable.

Analytical results are presented as "nondetects" ("U" qualifier) whenever chemical concentrations in samples do not exceed the detection or quantitation limits for the analytical procedures for those samples. Generally, the detection limit is the lowest concentration of a chemical that can be "seen" above the normal, random noise of an analytical instrument or method. To apply the previously mentioned statistical procedures to a data set with nondetects, a concentration value must be assigned to nondetects. Nondetects are assumed to be present at one-half the sample quantitation limit (SQL) (EPA, 1989a), although judgement is used in those cases where matrix interference or other phenomena drive the SQL unusually high.

The UCL or MDC, whichever is smaller, is selected as the source-term concentration, and is understood to represent a conservative estimate of average for use in the RA or in various transport models used to estimate exposure-point concentrations. If the data set consists of fewer than five data points, the MDC is selected as the source-term concentration.

2.2.2 Groundwater

Potential groundwater contamination is evaluated under two different scenarios as described in the following subsections.

Current Groundwater Conditions. Current concentrations of chemicals in groundwater from the available analytical data are evaluated in the RA. Applying the previously described UCL method to all the analytical data available for groundwater may underestimate the concentration to which a receptor may be exposed. The data set may contain data from wells at the periphery or outside the contamination plume. A more conservative and defensible approach is to estimate the source-term concentration as the arithmetic mean of the concentrations from the most contaminated part of the plume. Each COPC data set is evaluated individually to identify the data to be averaged to estimate the source-term concentration. Concentrations that exceed their background screening criterion and RBSC are included in the most contaminated part of the plume.

As previously noted, analytical data for current groundwater conditions will not be available for the RA. A groundwater investigation that will address the entire facility will be performed at a later date.

Future Groundwater Conditions. The future groundwater conditions scenario is created to evaluate the potential for chemicals in soil to leach to groundwater. Whether to evaluate the future groundwater conditions is a matter of professional judgement based on the nature and the concentrations of the contaminants identified in the soil and the groundwater. If the data suggest that the current groundwater conditions evaluation captures the potential for future contamination, the future groundwater conditions scenario is not evaluated. Reasons to decide against evaluating future groundwater conditions include:

- Releases to the soil are old; it is likely that the majority of the leaching has already occurred.
- Contaminants released to soil exhibit low mobility in the environment and, in the absence of the release of other solvents, are not expected to migrate to groundwater.
- Geological features render migration to groundwater unlikely or are expected to significantly slow the process.
- Concentrations in soil and groundwater are relatively low.
- Repeated sampling reveals concentrations in groundwater decreasing with time.

Reasons to decide for evaluating future groundwater conditions include:

- Releases to the soil are recent; it is likely that the majority of the leaching has yet to occur.
- Contaminants released to soil are generally mobile in the environment, or concurrent or subsequent releases of solvents might enhance migration to groundwater.
- Geological features do not hinder migration to groundwater.
- Concentrations in soil and groundwater are relatively high.
- Repeated sampling reveals concentrations in groundwater increasing with time.

If future groundwater conditions are evaluated, source-term concentrations of chemicals in soil are estimated as described above for soils. The source-term concentrations are then screened against the generic soil screening levels (SSL) based on a dilution-attenuation factor (DAF) of 20 (EPA, 1996). The DAF of 20 reflects the natural processes that reduce contaminant concentrations in the subsurface soil and groundwater for a 0.5-acre site. Sensitivity analysis, however, reveals that site size has little impact on the magnitude of the SSL, with slightly lower SSLs estimated for smaller site sizes (e.g., SSLs developed for a 0.5 acre area are approximately three-fold lower than those developed for a 30-acre site [EPA, 1996]). The Red Water Ponds Areas appear to be larger than 0.5 acres (IT, 1996); therefore, the SSLs based on a DAF of 20 are sufficiently conservative. Chemicals whose concentrations do not exceed the SSLs are not carried forth to a modeling exercise. Chemicals whose concentrations exceed the SSLs, and chemicals for which SSLs are not available, are carried forth to the modeling step and groundwater concentrations from leaching are modeled. These groundwater concentrations are considered to be the source-term concentrations under future groundwater conditions.

The modeled groundwater source-term concentrations are then subjected to risk-based screening, as previously described, to select groundwater COPC for future groundwater conditions. The future groundwater conditions COPC are used in the RA in the same way as the current groundwater conditions COPC.

2.3 The Data Summary Table

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

- Chemical name
- Frequency of detection
- Range of detected concentrations
- Range of detection limits
- Arithmetic mean of site concentrations
- UCL on the arithmetic mean
- Source-term concentration
- Appropriate RBSC
- Background screening criterion
- Selection as COPC
- Designation as a background COPC.

Footnotes in the tables provide the rationale for rejection of a chemical as a COPC.

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 COPC 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.

3.1 Conceptual Site Exposure Model

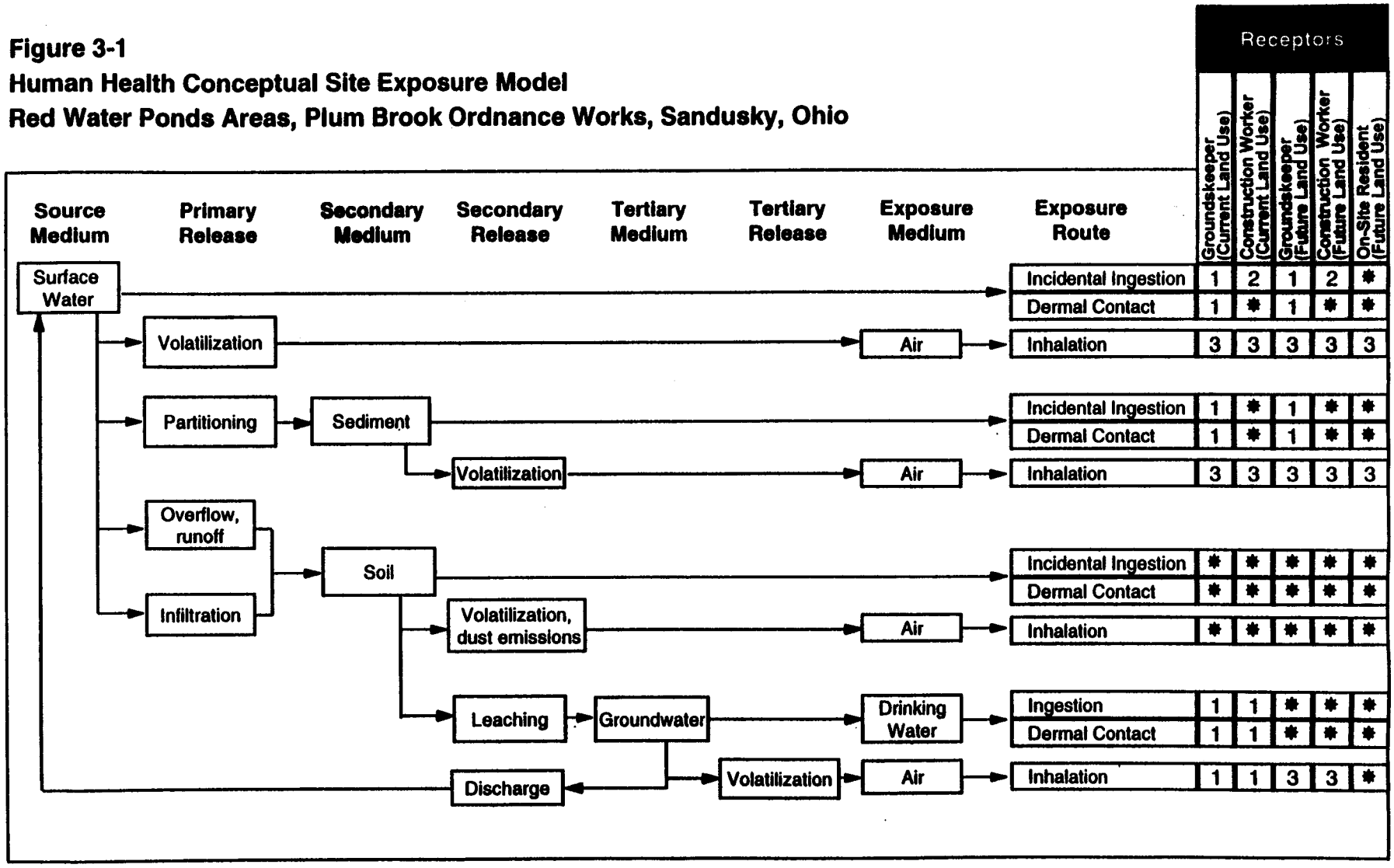
The conceptual site exposure model (CSEM) provides the basis for identifying and evaluating the potential risks to human health in the baseline RA. The CSEM (Figure 3-1) includes the receptors appropriate to all plausible land-use scenarios, and the potential exposure pathways. Graphically presenting 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 ensure that potential pathways are not overlooked. The elements of a CSEM include:

- Source (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.

The receptors and pathways in Figure 3-1 reflect plausible scenarios developed from information regarding site background and history, topography, climate, and demographics as presented in the scope of work (USACE, 1997) and the site-wide groundwater investigation (IT, 1997). Asterisks identify exposure pathways that are complete and addressed in the RA. Justification for

Figure 3-1
Human Health Conceptual Site Exposure Model
Red Water Ponds Areas, Plum Brook Ordnance Works, Sandusky, Ohio



* = Complete exposure pathway evaluated in the risk assessment.
1 = Incomplete exposure pathway.
2 = Incidential ingestion of surface water is expected to be insiginficant compared with dermal exposure.
3 = Although theoretically complete, large dilution factor of ambient air obviates the need to quantify this pathway.

exclusion of other pathways is provided in the footnotes. The CSEM is subject to revision as site investigation continues.

3.1.1 Physical Setting

The description of the physical setting may be brief, and may reference an earlier chapter or document where details may be found. Sufficient detail must be provided in the RA chapter or document to validate the selection of contaminated source, transport and exposure media, and to support the current and future land use and receptor scenarios selected for evaluation. The physical setting of the Red Water Ponds Areas is briefly summarized in Chapter 1.0. Although brief, the description supports the receptors and exposure pathways illustrated in Figure 3-1. Greater detail is provided in USACE (1997) and IT (1997).

3.1.2 Contaminant Sources, Release Mechanisms, and Migration Pathways

Contaminant sources, release mechanisms, and migration pathways are presented in Figure 3-1. Briefly, red water from the TNT manufacturing process was pumped to the Red Water Ponds. Contaminants in water may partition to sediment. Overflow and runoff may have spread contamination over the surrounding surface soil. Infiltration may have carried contaminants into the subsurface soil or groundwater. Contaminated shallow groundwater may discharge to the surface water, returning contaminants to the Red Water Ponds.

3.1.3 Receptors and Exposure Pathways

Receptors, selected to represent the upper bound on exposure from all plausibly exposed groups of people at the red water areas, and the pathways by which they may be exposed, are summarized in Figure 3-1 and Table 3-1. The exposure variable values used in the contaminant intake models are compiled in Table 3-2.

Most RAs 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, 1991). It is interpreted as reflecting the 90 to 95th percentile on exposure. In keeping with EPA (1991) guidance, variables chosen for a baseline RME scenario for intake rate, exposure frequency (EF) and exposure duration (ED) are generally upper bounds. Other variables, e.g., body weight (BW) and exposed skin surface area (SA), 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

Table 3-1

Receptor/Exposure Scenarios
Red Water Ponds Areas
Plum Brook Ordnance Works, Sandusky, Ohio

(Page 1 of 4)

Source Medium	Model	Exposure Medium	Exposure Pathway
CURRENT LAND USE			
Groundskeeper			
Surface soil	None	Soil	Incidental ingestion
			Dermal contact
	Volatilization from soil	Air	Inhalation
	Dust emissions based on activity	Air	Inhalation
Subsurface soil	Not quantified[a]		
Surface water	Not quantified[b]		
Sediment	Not quantified[b]		
Groundwater	Not quantified[c]		
Construction Worker			
Surface water	None	Surface water	Incidental ingestion: not quantified[d]
			Dermal contact
	Volatilization from water	Air	Inhalation: not quantified[e]
Sediment	None	Sediment	Incidental ingestion
			Dermal contact
	Volatilization from sediment	Air	Inhalation: not quantified[e]
Surface and subsurface soil	None	Soil	Incidental ingestion
			Dermal contact
	Volatilization from soil	Air	Inhalation
	Dust emissions based on activity	Air	Inhalation
Groundwater	Not quantified[c]		

Table 3-1

Receptor/Exposure Scenarios
Red Water Ponds Areas
Plum Brook Ordnance Works, Sandusky, Ohio

(Page 2 of 4)

Source Medium	Model	Exposure Medium	Exposure Pathway
FUTURE LAND USE			
Groundskeeper			
Surface soil	None	Soil	Incidental ingestion
			Dermal contact
	Volatilization from soil	Air	Inhalation
	Dust emissions based on activity	Air	Inhalation
Groundwater	None	Potable water	Drinking water ingestion
			Dermal contact
	Volatilization from water	Air	Inhalation: not quantified[e]
Subsurface soil	Not quantified[a]		
Surface water	Not quantified[b]		
Sediment	Not quantified[b]		
Construction Worker			
Surface water	None	Surface water	Incidental Ingestion: not quantified[d]
			Dermal contact
	Volatilization from water	Air	Inhalation: not quantified[e]
Sediment	None	Sediment	Incidental ingestion
			Dermal contact
	Volatilization from sediment	Air	Inhalation: not quantified[e]

Table 3-1

Receptor/Exposure Scenarios
Red Water Ponds Areas
Plum Brook Ordnance Works, Sandusky, Ohio

(Page 3 of 4)

Source Medium	Model	Exposure Medium	Exposure Pathway
Construction Worker (cont.)			
Surface and subsurface soil	None	Soil	Incidental ingestion
			Dermal contact
	Volatilization from soil	Air	Inhalation
	Dust emissions based on activity	Air	Inhalation
Groundwater	None	Potable water	Drinking water ingestion
			Dermal contact
	Volatilization from water	Air	Inhalation: not quantified[e]
On-Site Resident			
Surface and subsurface soil	None	Soil	Incidental ingestion
			Dermal contact
	Volatilization from soil	Air	Inhalation
	Dust emissions based on wind erosion	Air	Inhalation[f]
Groundwater	None	Potable water	Drinking water ingestion
			Dermal contact
	Volatilization from water[g]	Air	Inhalation
Surface water	None	Surface water	Incidental ingestion
			Dermal contact
	Volatilization from water	Air	Inhalation: not quantified[e]

Table 3-1

Receptor/Exposure Scenarios
Red Water Ponds Areas
Plum Brook Ordnance Works, Sandusky, Ohio

(Page 4 of 4)

Source Medium	Model	Exposure Medium	Exposure Pathway
Sediment	None	Sediment	Incidental ingestion
			Dermal contact
	Volatilization from sediment	Air	Inhalation: not quantified[e]

[a]Subsurface soil is covered with surface soil; there is no plausible mechanism for contact with this medium under the current site-use scenario.
[b]Although contact with this medium is possible, exposure would be sporadic, rather than continuous or predictable.
Such exposures do not lend themselves to evaluation under the chronic toxicity paradigm used in a baseline risk assessment.
[c]Groundwater is not developed as a potential source of potable water under the current site use scenario; there is no plausible mechanism for exposure to this medium.
[d]Although theoretically complete, incidental ingestion of surface water is expected to be insignificant relative to dermal exposure.
[e]Although theoretically complete, this pathway is not quantified because large volume of ambient air would effectively dilute concentrations of volatilized organic compounds to toxicologically insignificant levels.
[f]It is assumed that residential soil is 80 percent covered with pavement or vegetation for evaluating inhalation exposure via dust emissions.
[g]"Whole house" household-water-to-air volatilization model, adapted from information provided by U.S. Environmental Protection Agency (EPA), 1997, *Exposure Factors Handbook*, Final, National Center for Environmental Assessment, Washington, DC, EPA/600/P-95/002Fa, August.

Table 3-2

Variables Used to Estimate Potential Chemical Intakes and Contact Rates for Receptors Plum Brook Ordnance Works, Sandusky, Ohio

(Page 1 of 4)

Pathway Variable	Groundskeeper	Construction Worker	On-Site Resident
General Variables Used in All Intake Models			
Exposure duration (ED), years	25[a]	0.5[b]	Child: 6[c] Adult: 24[c]
Body weight (BW), kg	70[a]	70[a]	Child: 15[c] Adult: 70[a]
Averaging time, noncancer (AT), days[d]	9125	183	Child: 2190 Adult: 8760
Averaging time, cancer (AT), days[e]	25550	25550	Adult: 25550
Inhalation of VOCs and Resuspended Dust from Soil, or VOCs from Groundwater			
Fraction exposed to contaminated medium (FI_a), unitless	1[b]	1[b]	1[b]
Inhalation rate (IR_a), m^3/day	20[a]	20[a]	Child: 10[c] Adult: 20[a]
Exposure frequency (EF), days/year	250[a]	250[a]	350[a]
Incidental Ingestion of Soil			
Fraction exposed to contaminated medium (FI_{so}), unitless	1[b]	0.75[b]	0.9[b]
Soil incidental ingestion rate (IR_{so}), mg/day	100[a]	290[b]	Child: 200[a] Adult: 100[a]
Exposure frequency (EF), days/year	250[a]	250[a]	350[a]
Incidental Ingestion of Sediment			
Fraction exposed to contaminated medium (FI_{sd}), unitless	NA	0.25[b]	0.1[b]
Sediment incidental ingestion rate (IR_{sd}), mg/day	NA	290[b]	Child: 200[a] Adult: 100[a]
Exposure frequency (EF), days/year	NA	250[a]	52[b]

Table 3-2

Variables Used to Estimate Potential Chemical Intakes and Contact Rates for Receptors Plum Brook Ordnance Works, Sandusky, Ohio

(Page 2 of 4)

Pathway Variable	Groundskeeper	Construction Worker	On-Site Resident
Dermal Contact with Soil			
Fraction exposed to contaminated medium (FI_{so}), unitless	1[b]	0.75[b]	0.9[b]
Body surface area exposed to soil (SA_{so}), cm^2	11,300[f]	11,300[f]	Child: 1750[g] Adult: 4500[g]
Soil-to-skin adherence factor (AF_{so}), mg/cm^2	0.009[f]	0.08[f]	0.2[g]
Dermal absorption factor (ABS), unitless	csv	csv	csv
Exposure frequency (EF), days/year	250[a]	250[a]	350[a]
Dermal Contact with Sediment			
Fraction exposed to contaminated medium (FI_{sd}), unitless	NA	0.25[b]	0.1[b]
Body surface area exposed to sediment (SA_{sd}), cm^2	NA	11,300[f]	Child: 1750[g] Adult: 4500[g]
Sediment-to-skin adherence factor (AF_{sd}), mg/cm^2	NA	0.08[f]	0.2[g]
Dermal absorption factor (ABS), unitless	NA	csv	csv
Exposure frequency (EF), days/year	NA	250[a]	52[b]
Incidental Ingestion of Surface Water			
Fraction exposed to contaminated medium (FI_{sw}), unitless	NA	NA	1[b]
Incidental surface water ingestion rate, swimming scenario (IR_{sw}), L/hour	NA	NA	0.050[h]
Exposure time (ET_{sw}), hours/day	NA	NA	3[b]
Exposure frequency (EF), days/year	NA	NA	52[b]

Table 3-2

Variables Used to Estimate Potential Chemical Intakes and Contact Rates for Receptors Plum Brook Ordnance Works, Sandusky, Ohio

(Page 3 of 4)

Pathway Variable	Groundskeeper	Construction Worker	On-Site Resident
Dermal Contact with Surface Water			
Fraction exposed to contaminated medium (FI_{sw}), unitless	NA	1[b]	1[b]
Body surface area exposed to surface water, (SA_{sw}), cm^2	NA	3100[g]	Child: 7000[g] Adult: 18,150[g]
Permeability coefficient (PC), cm/hour	NA	csv	csv
Exposure time (ET_{sw}), hour/day	NA	2[b]	3[b]
Exposure frequency (EF), days/year	NA	250[a]	52[b]
Drinking Water Ingestion of Groundwater			
Fraction exposed to contaminated medium (FI_{gw}), unitless	1[b]	1[b]	1[b]
Drinking water ingestion rate (IR_{gw}), L/day	1[a]	1[a]	Child: 1[c] Adult: 2[a]
Exposure frequency (EF), days/year	250[a]	250[a]	350[a]
Dermal Contact with Groundwater			
Fraction exposed to contaminated medium (FI_{gw}), unitless	1[b]	1[b]	1[b]
Body surface area exposed to water (SA_{gw}), cm^2	4100[g]	4100[g]	Child: 7000[g] Adult: 18,150[g]
Permeability coefficient (PC), cm/hour	csv	csv	csv
Exposure time (ET_{gw}), hour/day	1[b]	1[b]	0.25[g]
Exposure frequency (EF), days/year	250[a]	250[a]	350[a]

[a]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.
[b]Assumed; see text.
[c]U.S. Environmental Protection Agency (EPA), 1998, *EPA Region 9: Preliminary Remediation Goals (PRGs) 1998*, May 1, on-line.
[d]Calculated as the product of ED (years) x 365 days/year.
[e]Calculated as the product of 70 years (assumed human lifetime) x 365 days/year.
[f]U.S. Environmental Protection Agency (EPA), 1997, *Exposure Factors Handbook*, Final, National Center for Environmental Assessment, Washington, DC, EPA/600/P-95/002Fa, August.

Table 3-2

Variables Used to Estimate Potential Chemical Intakes and Contact Rates for Receptors Plum Brook Ordnance Works, Sandusky, Ohio

(Page 4 of 4)

[g]EPA, 1992, *Dermal Exposure Assessment: Principles and Applications*, Interim Report, Office of Research and Development, Washington, DC, EPA/600/8-91/011B, including Supplemental Guidance dated August 18, 1992.
[h]U.S. Environmental Protection Agency (EPA), 1989, *Risk Assessment Guidance for Superfund*, Volume I, Human Health Evaluation Manual (Part A), Interim Final, Office of Emergency and Remedial Response, Washington, DC., EPA/540/1-89/002.

NA = not applicable to this receptor; csv = chemical-specific value.

dermal absorption factor (ABS) and soil-to-skin adherence factor (AF) for soil, and permeability coefficient (PC) and exposure time (ET) for water, only one variable, ABS or PC needs to be an upper bound. The conservatism built into the individual variables ensures that the entire estimate for contact rate is more than sufficiently conservative.

The averaging time (AT) for noncancer evaluation is computed as the product of ED (years) times 365 days/year, to estimate an average daily dose over the entire exposure period (EPA, 1989a). For cancer evaluation, AT is computed as the product of 70 years, the assumed human lifetime, times 365 days/year, to estimate an average daily dose prorated over a lifetime, regardless of the frequency or duration of exposure. This methodology 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 consistent with current EPA (1986) policy of carcinogen evaluation, although it introduces considerable uncertainty into the cancer RA.

A fractional term (FI) is introduced into the chemical intake equations to account for scenarios in which exposure to a potentially contaminated medium associated with the site is less than total daily exposure to that medium. For example, if the site of interest is small, so that a groundskeeper may spend only one-half of his working time at the site, an FI of 0.5 is applied to the soil ingestion and dermal intake equations. An FI is used also if a receptor's exposure is split between two comparable media. For example, if a construction worker is exposed to both soil and pond sediment, FIs are introduced that apportion his exposure between the two media. The default value of FI is 1.

Groundskeeper. The groundskeeper scenario is designed to evaluate the upper bound for site worker exposure to surface soil in the current and future (industrial) land-use scenario. Relevant exposure pathways include incidental ingestion and dermal contact. Inhalation of dust, raised by operating lawn mowers or other equipment, is also evaluated because relatively high dust concentrations may be produced within the groundskeeper's breathing zone, with little opportunity for dilution by the large volume of ambient air. Inhalation of VOC emissions from surface soil is a potentially complete exposure pathway that is also evaluated.

It is plausible that groundwater could be developed as a source of potable water under the future land-use scenario; therefore, contact with groundwater is also evaluated. Relevant pathways

include drinking water ingestion and dermal contact. Inhalation of VOC emissions from groundwater is also possible, but the large volume of ambient air is assumed to dilute airborne concentrations to toxicologically insignificant levels. In addition, groundwater usage time is relatively short, which reduces further the significance of the inhalation pathway. For these reasons, inhalation of VOC emissions from groundwater is not evaluated for the groundskeeper. It is assumed that contact with surface water or sediment, if it occurred at all, would be infrequent, and groundskeeper exposure to these media is not considered.

The groundskeeper is assumed to be a 70-kilogram (kg) adult who works 8 hours/day, approximately 5 days/week year-round on site for a total of 250 days/year for 25 years (EPA, 1991). The respiratory rate for the groundskeeper is assumed to be 20 cubic meters (m^3)/8-hour workday (2.5 m^3/hour), and the soil incidental ingestion rate is assumed to be 100 milligrams per day (mg/day), comparable to that for an agricultural worker.

The extent of soil adhesion to skin is surrounded with considerable uncertainty. EPA (1992b) reviewed the data available at the time and recommended 0.2 milligrams per square centimeter (mg/cm^2) as an average value and 1.0 mg/cm^2 as an upper bound for AF, regardless of the nature of the activity resulting in dermal exposure. However, it is probably more appropriate to use the average value for AF, because ABS, the other component of dermal uptake rate, is selected to represent an upper bound.

More recent studies evaluating soil adherence that consider the nature of the activity performed and the different body regions was reviewed by EPA (1997a). Measurements of soil adherence to hands, arms, legs, feet, and face for 29 groundskeepers revealed AFs ranging from 8E-4 mg/cm^2 (legs) to 1.5E-1 (hands). The AF weight-averaged across these body regions (i.e., adjusted to reflect the different surface areas of the different body regions) for males and females is 9E-3 mg/cm^2, which is used in this evaluation. The body regions evaluated for groundskeepers include approximately 11,300 cm^2 (EPA, 1997a).

In the future scenario, the groundskeeper may be exposed to groundwater, which could be developed as a source of drinking water. His drinking water ingestion rate is assumed to be 1 L/day (EPA, 1991). He may also experience dermal contact with groundwater used for irrigation, to clean equipment and to rinse dust or perspiration from his body. For this evaluation

it is assumed that the head, arms and hands, approximately 4,100 cm^2 (EPA, 1992b) are exposed intermittently throughout the day for up to an hour per day.

Construction Worker. The construction worker scenario is created to evaluate short-term exposure to subsurface as well as surface soil in either the current or future land-use scenario. Relevant exposure pathways include incidental ingestion and dermal contact. Inhalation of VOCs and dust raised by operating construction equipment is also evaluated. Unlike the groundskeeper scenario, inhalation of VOCs is evaluated. Construction activity may uncover significant subsurface soil concentrations of VOCs, facilitating emission of the VOCs directly into the construction worker's breathing zone.

It is plausible that groundwater could be developed as a source of potable water under the future land-use scenario; therefore, contact with groundwater is also evaluated. Relevant pathways are the same as described for the groundskeeper. In addition, exposure to surface water and sediment is plausible, as construction may involve draining, dredging, enlarging or moving the boundaries of the ponds. Dermal contact is expected to be the most important pathway for exposure to surface water and is evaluated. Incidental ingestion is possible, but is not expected to be significant relative to dermal contact and is not evaluated. For exposure to sediment, dermal contact and incidental ingestion may be important and both are evaluated. Inhalation of VOC emissions from surface water and sediment is possible, but the large volume of outdoor air is expected to dilute airborne concentrations to toxicologically insignificant levels, and this pathway is not evaluated.

The construction worker is assumed to be a 70-kg adult who works 8 hours/day, approximately 5 days/week year-round on site for a total of 250 days/year (EPA, 1991). Construction projects are assumed to last 6 months. The respiratory rate for the construction worker is assumed to be 20 m^3/8-hour workday (2.5 m^3/hour). Excavation and soil grading activities, which result in intensive soil contact, are assumed to last for 3 months; for the remaining 3 months, construction activities are assumed to result in less intensive soil contact. Soil and sediment ingestion rates of 480 and 100 mg/day are assumed for the intensive and less intensive soil contact periods, respectively resulting in a time-weighted average rounded to 290 mg/day.

It is assumed that the construction worker has less frequent contact with sediment than with soil. EPA (1989a) permits the development of a fraction to apportion his total daily exposure time

between the two media. It is assumed that three-fourths of the construction worker's exposure time involves contact with soil and one-fourth involves contact with sediment and surface water. Therefore, a fraction of 0.75 is introduced into the soil contact equations and a fraction of 0.25 is introduced into the sediment contact equations to estimate intake or contact rates. An exposure time of 2 hours/day is assumed for dermal contact with surface water.

An AF for a construction worker of 8E-2 mg/cm^2 is estimated as previously described for the groundskeeper, combining EPA (1997a) data for construction workers, utility workers, and equipment operators to capture the full range of activities likely to be performed by this receptor. The body regions evaluated for construction workers include approximately 11,300 cm^2. It is assumed that dermal exposure to surface water involves only the hands and arms, with a body surface area estimated at 3,100 cm^2 (EPA, 1992b).

In the future scenario, the construction worker may be exposed to groundwater, which could be developed as a source of drinking water. His drinking water ingestion rate is assumed to be 1 L/day (EPA, 1991). 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, arms and hands, approximately 4,100 cm^2 (EPA, 1992b) are exposed intermittently throughout the day for up to an hour/day.

On-Site Resident. The on-site resident scenario is created to evaluate the upper bound for exposure to site surface water, sediment, soil, and groundwater under the future land-use scenario. Relevant pathways for soil exposure include incidental ingestion, dermal contact, and inhalation of VOCs and dust. For evaluating inhalation exposure to dust, it is assumed that 80 percent of the surface is covered with pavement or vegetation. Relevant pathways for groundwater exposure include drinking water ingestion, dermal contact, and inhalation of VOCs emitted during household use of water.

The resident may also have access to the ponds, thereby becoming exposed to surface water and sediment. It is assumed that weather conditions are such that he spends 8 hours/day, 2 day/week, at the ponds during the warmer half of the year (i.e., 52 days/year). He is assumed to swim, wade, or play in the ponds 3 hours/day. Incidental ingestion of surface water during water play is assumed to be 0.050 L/hour (EPA, 1989a). Direct contact with sediment occurs during the 8 hours/day spent in or near the ponds.

Inhalation of airborne VOCs emitted from surface water or sediment is a potentially complete pathway. It is assumed, however, that the large volume of outdoor air would effectively dilute airborne concentrations to toxicologically insignificant levels, and the inhalation pathway is not quantified.

The on-site residential scenario is evaluated using both an adult and child. Cancer risk is estimated as the sum of the risks calculated for the adult and the child. The child is used for the noncancer evaluation. This approach captures the greater conservatism of the larger incidental soil ingestion and drinking water ingestion rates for the child, when normalized for body weight.

The adult resident is assumed to be a 70-kg person with an incidental soil/sediment ingestion rate of 100 mg/day, an inhalation rate of 20 m^3/day (0.83 m^3/hour), and a drinking water ingestion rate of 2 L/day (EPA, 1991). His entire body surface area, approximately 18,150 cm^2 (EPA, 1992b), is available for contact with water while bathing or swimming. Body surface area exposed to soil or sediment is assumed to be 25 percent of total surface area, or approximately 4,550 cm^2 (EPA, 1992b). The adult resident is assumed to spend 0.25 hours/day bathing (EPA, 1992b). The adult resident is exposed to soil and groundwater 350 days/year for 24 years (EPA, 1998a).

The child resident is assumed to be a 1- through 6 year-old with an average body weight of 15 kg, a soil/sediment ingestion rate of 200 mg/day, an inhalation rate of 10 m^3/day (0.42 m^3/hour) and a drinking water ingestion rate of 1 L/day (EPA, 1997a). His entire body surface area, approximately 7,000 cm^2, is available for exposure to surface water while swimming or playing in water, or to groundwater during a 0.25 hour per day bath (EPA, 1992b). Body surface area exposed to soil or sediment is assumed to be 25 percent of total surface area, or approximately 1,750 cm^2 (EPA, 1992b). The child receptor is exposed to soil and groundwater for 350 days/year for 6 years.

EPA (1989a) permits the development of a fraction to reflect the proportion of total daily exposure that a receptor obtains from potentially contaminated medium. In this scenario, the fraction term is used to apportion the resident's time of exposure between site soil and sediment. It is assumed that the resident spends 16 hours/day awake and potentially exposed to soil or sediment. As previously noted, 350 days/year are available for contact with soil; 52 of those days are also available for contact with sediment. It is assumed that contact with soil and sediment

does not occur simultaneously; i.e., on those days when the resident spends time at the pond, 8 hours are spent in contact with surface soil and 8 hours are spent in contact with sediment. The fraction of exposure to soil, therefore, is 16 hours/16 hours = 1 on the 298 days without time spent at the pond, and 8 hours/16 hours = 0.5 on the 52 days with some time spent at the pond. An overall weighted fraction of 0.93 (rounded to 0.9) is estimated for exposure to soil. An overall weighted fraction of 0.07 (rounded to 0.1) is estimated for exposure to sediment.

An average soil and sediment AF of 0.2 mg/cm^2 (EPA, 1992b) is used in this evaluation.

Other Receptors Not Considered. Other plausible site workers include office workers and delivery personnel. These workers, however, would be less intensively exposed to soil or groundwater than the groundskeeper; therefore, their exposures are not evaluated. The Red Water Ponds Areas could become part of the area used for National Guard training activities. National Guard trainees, however, would be less exposed to any of the potentially contaminated media than the previously described receptors; their exposures are not evaluated. Parts of PBOW are used for fishing and hunting. The Red Water Ponds, however, are not used for fishing, and the Red Water Ponds Areas are too small to provide a significant amount of browse for deer or food for other game animals that would make a significant contribution to the diet. Furthermore, experience has shown that the game ingestion pathway generally is insignificant for contaminants other than the PCDDs/PCDFs, and the previously described receptors would represent the upper bound for exposure to soil, surface water, and sediment. For these reasons, a hunting scenario is not evaluated. Much of the land around PBOW is used for agriculture, and it is possible that areas of PBOW could be farmed in the future. The areas around the Red Water Ponds, however, are not ideally suited for agriculture. Also, experience has shown that ingestion of home-grown grains, fruits, meat, poultry, eggs, or milk is generally insignificant compared with direct exposure pathways, except for PCDDs/PCDFs; therefore, indirect food-chain exposures are not evaluated.

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

The last receptor considered is a trespasser or site visitor who would use the site for recreational purposes. This scenario, however, is not plausible under the current site-use scenario, because

the facility is fenced and patrolled. The site visitor scenario is plausible under the future site-use scenario (the fence and patrols may be removed in the future), but the resident would represent the upper bound on exposure, and evaluation of a site visitor is not necessary.

3.2 Quantification of Exposure-Point Concentrations

The exposure-point concentrations of COPC for direct exposure pathways for soil, surface water, and sediment will be the source-term concentrations estimated as described in Section 2.2. The exposure-point concentrations of COPC in groundwater for all evaluations except dermal uptake of VOCs for the adult resident will be the source-term concentrations estimated as described in Section 2.2. The resident is evaluated for inhalation exposure to VOCs volatilizing from groundwater during household use. Volatilization effectively reduces the concentration of VOCs remaining in the water available for dermal uptake. Estimation of exposure-point concentrations of VOCs in groundwater for dermal uptake for the resident is described in Section 3.2.2.

3.2.1 Exposure-Point Concentrations in Air

3.2.1.1 COPC Concentrations from Dust

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

$$C_a = (D)(C_{so})(CF1) \qquad \text{Eq. 3.1}$$

where:

C_a = contaminant concentration in air (mg/m^3, calculated)
D = dust loading factor (grams [g] of soil/m^3 of air)
C_{so} = contaminant concentration in soil (mg/kg)
CF1 = conversion factor (1E-3 kg/g).

Plausible values for D include 2E-4 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 is estimated for the construction worker.

The resident is more likely to be exposed to dust that arises from wind erosion rather 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:

$$PEF = Q/C \cdot \frac{3600}{0.036 \cdot (1 - V) \cdot (U_m/U_t)^3 \cdot F(x)} \qquad \text{Eq. 3.2}$$

where:

PEF = particulate emission factor (m^3/kg, calculated)
Q/C = inverse of the mean concentration at center of square source (50.38 g/m^2-second per kg/m^3, site-specific value from Table 3 in EPA, 1996 [Zone 5, Cleveland, 10-acre site])
3600 = seconds/hour
V = fraction of surface covered with vegetation (0.8, unitless, assumed)
U_m = mean annual wind speed (default 4.69 m/second)
U_t = equivalent threshold value of wind speed at 7 m (default 11.32 m/second)
F(x) = function dependent on U_m/U_t (default 0.194).

The concentration of COPC in air is calculated as follows:

$$C_a = \frac{C_{so}}{PEF} \qquad \text{Eq. 3.3}$$

where:

C_a = contaminant concentration in air (mg/m^3)
C_{so} = contaminant concentration in soil (mg/kg)
PEF = particulate emission factor (m^3/kg).

3.2.1.2 VOC Concentrations from Soil

The groundskeeper, construction worker, and resident may be exposed to VOCs released from soil by volatilization. Exposure-point concentrations 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):

$$VF_s = Q/C \cdot CF2 \cdot \left(\frac{[3.14 \cdot D_A \cdot T\ ]^{1/2}}{2 \cdot \rho_b \cdot D_A} \right) \qquad \text{Eq. 3.4}$$

and

$$D_A = \frac{(\theta_a^{10/3} \cdot D_i \cdot H' + \theta_w^{10/3} \cdot D_W)/n^2}{\rho_b \cdot K_d + \theta_w + \theta_a \cdot H'} \qquad \text{Eq. 3.5}$$

where:

VF_s = chemical-from-soil volatilization factor (m^3/kg, chemical-specific, calculated)
Q/C = inverse of the mean concentration at center of square source (50.38 g/m^2-second per kg/m^3, site-specific value from Table 3 of EPA, 1996 [Zone 5, Cleveland, 10-acre site)
CF2 = conversion factor (1E-4 m^2/cm^2)
D_A = apparent diffusivity (cm^2/second, calculated)
T = exposure interval (seconds, estimated as ED · 3.15E7 seconds/year)
ED = exposure duration (years, receptor-specific)
ρ_b = dry soil bulk density (1.5 g/cm^3, default, or site-specific)
θ_a = air-filled soil porosity (0.28 unitless, default, or site-specific estimated as n-θ_w)
n = total soil porosity (0.43 unitless, default, or site-specific estimated as 1-[ρ_b/ρ_s])
ρ_s = true soil or particle density (2.65 g/cm^3, default, or site-specific)
θ_w = water-filled soil porosity (0.15 L_{water}/L_{soil}, default, or site-specific)
D_i = diffusivity in air (cm^2/second, chemical specific)
H' = dimensionless Henry's law constant (chemical-specific, may be estimated as H · 41
H = Henry's law constant (atmosphere-m^3/mole, chemical-specific)
D_W = diffusivity in water (cm^2/second, chemical-specific)
K_d = soil-water partition coefficient (cm^3/g, chemical-specific, may be estimated as $K_{oc} \cdot f_{oc}$)
K_{oc} = soil organic carbon-water partition coefficient (cm^3/g, chemical-specific)
f_{oc} = organic carbon content of soil (6E-3 g/g, default, or site-specific).

The concentration of COPC in air is calculated by substituting VF_s for PEF in Equation 3.3.

3.2.1.3 VOC Concentrations from Groundwater

Inhalation of VOCs released from groundwater used as household water is evaluated for the resident. A model for the rate of VOC release from groundwater is used to estimate an average concentration of VOCs in air, as follows:

$$C_a = \frac{W_{house} \cdot K_m \cdot C_g \cdot t}{VR_{house} \cdot T} \qquad \text{Eq. 3.6}$$

where:

C_a	=	concentration of VOC in indoor air, average for entire house adjusted to reflect average time spent indoors (mg/m^3, calculated).
W_{house}	=	average daily household indoor water use (447 L/day, see below).
K_m	=	VOC mass transfer coefficient, all household uses, (0.54, unitless, see below)
C_g	=	concentration of VOC in groundwater (mg/L)
t	=	time spent by resident indoors (16.4 hours/day, EPA, 1997a)
VR_{house}	=	average daily ventilation rate (3984 m^3/day, see below).

EPA (1997a) reviewed several studies of the volumes of houses and air exchange rates, and recommends 369 m^3 as a central estimate of the volume of a house, and 0.45 air changes/hour as a typical air exchange rate. A ventilation rate for a typical house of 166 m^3/hour (3,984 m^3/day) is estimated from these values. EPA (1997a) also related residential volumes to household size. The average volume of a one-person household is 269 m^3; the average volume of a two-person household is 386 m^3. The average volume of all households, 369 m^3 as previously described, is similar to the volume for a two-person household. Therefore, it is assumed that the average household contains two persons. EPA (1997a) estimated a mean per capita indoor water use rate of 59 gallons/day. Assuming a two-person household, this estimate is equivalent to 118 gallons/day, or 447 L/day.

McKone and Knezovich (1991) measured the transfer of trichloroethene (as a typical VOC) from shower water to shower room air and obtained an average mass transfer coefficient of 0.6. They noted that this was similar to the value for radon (0.70 for showers and 0.54 for all household

uses of water). They also observed that when the dimensionless Henry's law constant is greater than 1E-1, mass transfer is almost completely dependent on the diffusion coefficient of the chemical in water; however, this parameter varies little for most organic chemicals in the 100 to 500 g/mole range. Therefore, the mass transfer coefficient for radon of 0.54 estimated for all household uses of water is used for all VOCs in this evaluation.

3.2.2 Exposure-Point Concentration of VOCs in Groundwater: Resident Dermal Uptake

As previously noted, volatilization of VOCs from household water reduces the concentration remaining available for dermal contact (it is assumed that groundwater used as drinking water is ingested before significant volatilization occurs). The concentration of VOCs remaining in the water are calculated by difference as follows:

$$C_d = C_g \cdot (1 - K_m) \qquad \text{Eq. 3.7}$$

where:

C_d = concentration of VOC in household water available for dermal exposure (mg/L, calculated)
C_g = concentration of VOC in groundwater (mg/L)
K_m = VOC mass transfer coefficient, all household uses, (0.54, unitless).

3.3 Quantification of Chemical Intake

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

3.3.1 Inhalation of COPC in Air

The following equation is used to estimate the inhaled dose of COPC in air (groundskeeper, construction worker, and resident: inhalation of dust and VOCs from soil, inhalation of VOCs from soil, inhalation of dust; resident adult and child: inhalation of VOCs from groundwater):

$$I_a = \frac{(C_a)(FI_a)(IR_a)(EF)(ED)}{(BW)(AT)} \qquad \text{Eq. 3.8}$$

where:

I_a = inhaled dose of COPC (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 COPC in Soil

The ingested dose of COPC in soil (groundskeeper, construction worker, resident adult and child) is estimated from the equation:

$$I_{so} = \frac{(C_{so})(FI_{so})(IR_{so})(EF)(ED)(CF4)}{(BW)(AT)} \qquad \text{Eq. 3.9}$$

where:

I_{so} = ingested dose of COPC in soil (mg/kg-day, calculated)
C_{so} = concentration of COPC in soil (mg/kg)
FI_{so} = fraction of exposure attributed to site soil (unitless)
IR_{so} = ingestion rate of soil (mg/day)
EF = exposure frequency (days/year)
ED = exposure duration (years)
CF4 = conversion factor (1E-6 kg/mg)
BW = body weight (kg)
AT = averaging time (days).

3.3.3 Incidental Ingestion of COPC in Sediment

The ingested dose of COPC in sediment (construction worker, adult and child resident) is estimated from the equation:

$$I_{sd} = \frac{(C_{sd})(FI_{sd})(IR_{sd})(EF)(ED)(CF4)}{(BW)(AT)} \qquad \text{Eq. 3.10}$$

where:

I_{sd} = ingested dose of COPC in sediment (mg/kg-day, calculated)
C_{sd} = concentration of COPC in sediment (mg/kg)
FI_{sd} = fraction of exposure attributed to site sediment (unitless)
IR_{sd} = ingestion rate of sediment (mg/day)
EF = exposure frequency (days/year)
ED = exposure duration (years)
CF4 = conversion factor (1E-6 kg/mg)
BW = body weight (kg)
AT = averaging time (days).

3.3.4 Incidental Ingestion of Surface Water

The ingested dose of COPC in surface water (resident adult and child) is estimated from the equation:

$$I_{sw} = \frac{(C_{sw})(FI_{sw})(IR_{sw})(ET_{sw})(EF)(ED)}{(BW)(AT)} \qquad \text{Eq. 3.11}$$

where:

I_{sw} = ingested dose of COPC in surface water (mg/kg-day, calculated)
C_{sw} = concentration of COPC in surface water (mg/L)
FI_{sw} = fraction of exposure attributed to site surface water (unitless)
IR_{sw} = ingestion rate of surface water (L/hour)
ET_{sw} = exposure time (hours/day)
EF = exposure frequency (days/year)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (days).

3.3.5 Drinking Water Ingestion of Groundwater

The ingested dose of COPC in groundwater (future scenario: groundskeeper, construction worker, on-site resident) is estimated from the following equation:

$$I_{gw} = \frac{(C_{gw})(FI_{gw})(IR_{gw})(EF)(ED)}{(BW)(AT)} \qquad \text{Eq. 3.12}$$

where:

I_{gw} = ingested dose of COPC in groundwater (mg/kg-day, calculated)
C_{gw} = concentration of COPC in groundwater (mg/L)
FI_{gw} = fraction of exposure attributed to site groundwater (unitless)
IR_{gw} = drinking water ingestion rate (L/day)
EF = exposure frequency (days/year)
ED = exposure duration (years)
BW = body weight (kg)
AT = averaging time (days).

3.3.6 Dermal Contact with COPC in Soil, Sediment, or Water

Unlike the methodologies for estimating inhaled or ingested dose of COPC, which quantify the dose presented to the barrier membrane (the pulmonary or gastrointestinal mucosa, respectively), dermal dose is estimated as the dose that crosses the skin and is systemically absorbed. For this reason, dermal toxicity values are also based on absorbed dose. The absorbed dose of COPC is estimated from the equation (EPA, 1992b):

$$DAD = \frac{(DA)(SA)(EF)(ED)}{(BW)(AT)} \qquad \text{Eq. 3.13}$$

where:

DAD = average dermally absorbed dose of COPC (mg/kg-day, calculated)
DA = dose absorbed per unit body surface area per day (mg/cm^2-day)
SA = SA_{so} for soil, SA_{sd} for sediment, SA_{sw} for surface water, 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 (DA) is calculated differently for dermal uptake from soil or sediment and from water. Dermal uptake of constituents from soil (groundskeeper, construction worker, resident) or sediment (construction worker, resident) assumes that absorption is a function of the fraction of a dermally applied dose that is absorbed. It is calculated from the equation (EPA, 1992b):

$$DA = (C)(FI)(CF4)(AF)(ABS) \qquad \text{Eq. 3.14}$$

where:

DA = dose absorbed per unit body surface area per day (mg/cm^2-day, calculated)
C = C_{so} for soil, C_{sd} for sediment, = concentration of COPC in medium (mg/kg)
FI = FI_{so} for soil, FI_{sd} for sediment, = fraction of exposure attributed to site medium (unitless)
CF4 = conversion factor (1E-6 kg/mg)
AF = AF_{so} for soil, AF_{sd} for sediment, = soil- or sediment-to-skin adherence factor (mg/cm^2-day)
ABS = absorption fraction (unitless, chemical-specific).

ABS values have been empirically determined for very few chemicals. EPA (1992b) discussed the available empirical data, as well as several predictive approaches for estimating ABS, but refrains from recommending any single approach. OEPA (1998) offers ABS default values of 10 percent (0.1) for organic chemicals and 1 percent (0.01) for inorganic chemicals, consistent with EPA (1998a). These default values will be used when empirical data are not available. The ABS values for soil will be used also for sediment.

Quantification of dermal uptake of COPC from surface water (construction worker, resident) and groundwater (groundskeeper, construction worker, resident) depends on a PC, which describes the rate of movement of a constituent from water across the dermal barrier to the systemic circulation (EPA, 1992b). The equation for dermal uptake of chemicals from water is the same as the equation for dermal uptake of chemicals from soil. DA for dermal uptake from water is calculated from the following equation:

$$DA = (C)(FI)(PC)(ET)(CF5) \qquad \text{Eq. 3.15}$$

where:

DA = dose absorbed per unit body surface area per day (mg/cm^2-day, calculated)
C = C_{sw} for surface water, C_{gw} for groundwater, = concentration of COPC in water (mg/L)
FI = FI_{sw} for surface water, FI_{gw} for groundwater, = fraction of exposure attributed to site medium (unitless)
PC = permeability coefficient (cm/hour)
ET = ET_{sw} for surface water, ET_{gw} for groundwater, = time of exposure (hours/day)
CF5 = conversion factor (1E-3 L/cm^3).

PC has been determined for very few inorganic compounds. For those inorganic compounds for which empirical data are not available, EPA (1992b) recommends a default of 1E-3 cm/hour.

PC for organic chemicals varies by several orders of magnitude. PC for organic chemicals is highly dependent on lipophilicity, expressed as a function of the octanol/water partition coefficient (K_{ow}).

When possible, values for PC and τ are taken from EPA (1992b). If PC values are not available, they will be calculated from the formula (EPA, 1992b):

$$Log(PC) = -2.72 + 0.71(\log K_{ow}) - 0.0061(MW) \qquad \text{Eq. 3.16}$$

where:

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

If values for τ are not available, they are calculated from the equation (EPA, 1992b):

$$\tau = \frac{L_{sc}}{6x10^{(-2.72-0.0061*MW)}} \qquad \text{Eq. 3.17}$$

where:

τ	=	time for concentration of contaminant in stratum corneum to reach steady state (hours, calculated)
L_{sc}	=	effective thickness of the stratum corneum (1E-3 cm)
MW	=	molecular weight.

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 Cancer Evaluation

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, 1986). The qualitative aspect is a weight-of-evidence evaluation of the likelihood that a chemical might induce cancer in humans. The EPA recognizes six weight-of-evidence group classifications for carcinogenicity:

- Group A - Human Carcinogen: human data are sufficient to identify the chemical as a human carcinogen
- Group B1 - Probable Human Carcinogen: human data indicate that a causal association is credible, but alternative explanations cannot be dismissed
- Group B2 - Probable Human Carcinogen: human data are insufficient to support a causal association, but testing data in animals support a causal association
- Group C - Possible Human Carcinogen: human data are inadequate or lacking, but animal data suggest a causal association, although the studies have deficiencies that limit interpretation
- Group D - Not Classifiable as to Human Carcinogenicity: human and animal data are lacking or inadequate
- Group E - Evidence of Noncarcinogenicity to Humans: human data are negative or lacking, and adequate animal data indicate no association with cancer.

The toxicity value for carcinogenicity, called a cancer slope factor (SF), is an estimate of potency. Potency estimates are developed only for chemicals in Groups A, B1, B2 and C, and only if the data are sufficient. The potency estimates 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 its effect 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 done by estimating an equivalent human dose from the animal dose.
- The range of sensitivities arising from genotypic and phenotypic diversity in the human population is not reflected in the animal models ordinarily used in cancer studies.
- Usually very high doses of chemical are used, which may alter normal biology, creating a physiologically artificial state and introducing substantial uncertainty regarding the extrapolation to the low-dose range expected with environmental exposure.

- Individual studies vary in quality (e.g., duration of exposure, group size, scope of evaluation, adequacy of control groups, appropriateness of dose range, absence of concurrent disease, sufficient long-term survival to detect tumors with long induction or latency periods).

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

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

where:

$p_{(d)}$ = the probability of cancer associated with dose = 1 mg/kg-day
$p_{(0)}$ = the background probability of developing cancer at dose = 0 mg/kg-day.

The SF is expressed as risk per mg/kg-day. In order to be appropriately conservative, the SF is usually the 95 percent upper bound on the slope of the dose-response curve extrapolated from high (experimental) doses to the low-dose range expected in environmental exposure scenarios. EPA (1986) assumes that there are no thresholds for carcinogenic expression; therefore, any exposure represents some quantifiable risk.

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

The EPA (1998b) Integrated Risk Information System (IRIS) expresses inhalation cancer potency as a unit risk factor (URF) based on concentration, or risk per microgram (µg) of chemical/m^3 of ambient air. Because cancer risk characterization requires a potency expressed as risk per mg/kg-day, the URF must be converted to the mathematical equivalent of an inhalation cancer SF, or risk per unit dose. Since the inhalation unit risk is based on continuous lifetime exposure of an adult human (assumed to inhale 20 m^3 of air/day and to weigh 70 kg) the mathematical conversion consists of multiplying the unit risk (per µg/m^3) by 70 kg and by 1,000 µg/mg, and dividing the result by 20 m^3/day.

4.2 Evaluation of Noncancer Effects

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

- Qualitative identification of the adverse effect(s) associated with the chemical; these may differ depending on the duration (e.g., acute or chronic) or route (e.g., 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; 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 data base, in regard to developing a reference dose (RfD) for human exposure
- Identification of the target organ 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 uncertainty factor.

IRIS (EPA, 1998b) and the Health Effects Assessment Summary Tables (HEAST) (EPA, 1997b) express 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. Since the inhalation RfC is based on continuous exposure of an adult human (assumed to inhale 20 m^3 of air/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.

4.3 Target Organ Toxicity

As a matter of science policy, EPA (1989a) assumes dose- and effect-additivity for noncarcinogenic effects. This assumption provides the justification for adding the hazard quotients (HQ) or HIs in the risk characterization for noncancer effects resulting from exposure to multiple chemicals, pathways or media. EPA, however, 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.

Mechanism 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, unless the data clearly indicate otherwise. 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.

As a practical matter, since 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 at the threshold, the more severely affected is chosen. 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.

4.4 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 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 SFs are expressed as reciprocal doses.

4.5 Sources of Toxicity Information Used in the Risk Assessment

Toxicity values are chosen using the following hierarchy:

- EPA's on-line IRIS database (EPA, 1998b) containing toxicity values that have undergone the most rigorous Agency review
- The latest version of the annual HEAST, including all supplements (EPA, 1997b)
- Other EPA documents, memoranda, former Environmental Criteria and Assessment Office, or National Center for Environmental Assessment (NCEA) derivations for the Superfund Technical Support Center.

All toxicity values, regardless of their source, are evaluated for appropriateness for use in RA.

When toxicity values are not located, the primary literature may be surveyed to determine whether sufficient data exist that would permit derivation of a toxicity value. The use of surrogate chemicals is also considered, if the chemical structure, adverse effects and toxic potency of the surrogate and chemical of interest are judged to be sufficiently similar. OEPA toxicologists will be consulted should it become necessary to develop toxicity values for any chemical.

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

- Oral absorption efficiency data compiled by the NCEA for the Superfund Health Risk Technical Support Center of the EPA
- Federal agency reviews of the empirical data, such as ATSDR 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 for deriving dermal toxicity values from oral toxicity values. The suitability of the GAF increases when the following similarities are present in the oral toxicokinetic 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 was 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.

The most defensible GAF for each chemical is used in the RA.

When quantitative data are insufficient, a default GAF is used. The EPA (1995) Region IV default GAFs of 0.8 for VOCs, 0.5 for SVOCs, and 0.2 for inorganic chemicals will be used in this evaluation.

Individual profiles are not provided for most of the COPC, because the toxicity information and its documentation presented in the summary tables are sufficient for quantitative risk characterization and uncertainty analysis. However, chemicals for which there are controversies or issues that would impact risk characterization are discussed herein.

5.0 Risk Characterization

Risk characterization is the combination of the results of the exposure assessment and toxicity assessment to yield a quantitative expression of cancer risk or noncancer hazard for the exposed receptors. This quantitative expression is the probability of developing cancer, or a non-probabilistic comparison of estimated dose with a reference dose for noncancer effects. Quantitative estimates are developed for individual chemicals, exposure pathways, and exposure media for each receptor. The risk characterization is used to guide risk management decisions.

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, are generally overly conservative, because risk characterization involves multiplication of the conservatisms built into the estimation of source-term and exposure-point concentrations, the exposure (intake) estimates and the toxicity dose-response assessments.

Risk characterization is limited to those chemicals selected as COPC, i.e., present at concentrations that exceed RBSCs (Section 2.1.4). The COPC are further categorized as background or site-related (Section 2.1.3). Three risk evaluations will be provided for each medium of interest: Total Site Risk, which includes all COPC identified in the medium; Background Risk, which includes the inorganic and organic COPC identified as natural or anthropogenic background; and Site-Related Risk, which is limited to COPC not identified as background and presumed to be present as a result of former PBOW activity.

5.1 Carcinogenic Effects of Chemicals

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 (per mg/kg-day).

The CDI term in Equation 5.1 is equivalent to the "I" or "DAD" terms (intake or dose) in Equations 3.8 through 3.13, 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:

$$ILCR = 1 - e^{[(CDI)(SF)]} \qquad \text{Eq. 5.2}$$

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 chemical's mechanisms of toxicity or sites (organs of the body) of action. Cancer risk arising from simultaneous exposure by a given pathway to multiple chemicals is estimated from the following equation:

$$Risk_p = ILCR_{(chem1)} + ILCR_{(chem2)} + \ldots ILCR_{(chemi)} \qquad \text{Eq. 5.3}$$

where:

$Risk_p$ = total pathway risk of cancer incidence, calculated
$ILCR(chem_i)$ = individual chemical cancer risk.

Cancer risk for a given receptor across pathways and across media is summed in the same manner.

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 (EPA, 1989a):

$$HQ = I/RfD \qquad \text{Eq. 5.4}$$

where:

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

The I term in Equation 5.4 is equivalent to the "I" or "DAD" terms (intake or dose) in Equations 3.8 through 3.13, 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 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 several chemicals, an HI is calculated as the sum of the HQs by:

$$HI = I_1/RfD_1 + I_2/RfD_2 + \ldots I_i/RfD_i \qquad \text{Eq. 5.5}$$

where:

HI = hazard index (unitless, calculated)
I_i = intake for the ith toxicant
RfD_i = reference dose for the ith toxicant.

If HI for a given pathway exceeds 1, individual HI values may be calculated for each target organ.

5.3 Risk-Based Remediation Criteria Development

RBRC development performed as part of the RA provides much useful material to risk managers. RBRC are site-specific risk-based concentrations that reflect the exposure and toxicity assumptions applied in the BLRA. Consequently the RBRC are source medium-, receptor-, and chemical-specific.

The first step in RBRC development is selection of COC. Either of two conditions result in designation of a COPC as a COC:

- The concentration of the COPC exceeds its medium-specific ARAR.
- The COPC contributes significantly to unacceptable cancer risk (total site ILCR greater than 1E-5) or hazard (total site HI greater than 1).

Significant contribution to cancer risk is defined as contributing an ILCR across all exposure pathways for a given source medium exceeding 1E-6; significant contribution to hazard is defined as contributing an HI across all exposure pathways for a given source medium exceeding 0.1. The COC, therefore, may be selected because of their cancer risk (cancer COC) or non-cancer hazard (noncancer COC). Care will be taken to ensure that the cumulative risk or HI for COPC eliminated from the list of COC does not exceed the acceptable limits.

RBRC are risk- or hazard-specific concentrations of chemicals developed only for the COC in media that are associated with unacceptable risk. RBRC for cancer COC are estimated for a given medium from the following equation:

$$RBRC_{coc} = \frac{ST_{coc}\,TR}{ILCR_{COC}} \qquad \text{Eq. 5.6}$$

where:

$RBRC_{coc}$ = remedial goal option for a given COC, receptor and source medium (calculated)

ST_{coc}	=	source-term concentration of the COC in the given medium
TR	=	target risk level (1E-6, 1E-5)
$ILCR_{coc}$	=	total incremental lifetime cancer risk for a given COC, receptor and source medium.

RBRC for noncancer COC are estimated as follows:

$$RBRC_{coc} = \frac{ST_{coc}\,THI}{HI_{COC}} \qquad \text{Eq. 5.7}$$

where:

$RBRC_{coc}$	=	remedial goal option for a given COC, receptor and source medium (calculated)
ST_{coc}	=	source-term concentration of the COC in the given medium
THI	=	target hazard index (0.1, 1)
HI_{coc}	=	total hazard index for a given COC, receptor and source medium.

Concentration units are not provided in Equations 5.6 or 5.7; the RBRC units will be the same as the concentration units of the source-term concentration.

6.0 Uncertainty Analysis

This section explores the uncertainties inherent in the RA process. Uncertainty is a factor in each step of the data evaluation and exposure and toxicity assessments presented in the preceding sections. Uncertainties associated with earlier stages of the RA become magnified when they are concatenated with other uncertainties in the latter stages. It is not possible to eliminate all uncertainty; however, a recognition of the uncertainties is fundamental to the understanding and reasonable use of the RA results.

Generally, RAs carry two types of uncertainty. Measurement uncertainty refers to the usual variance that accompanies scientific measurements, e.g., instrument uncertainty (accuracy and precision) associated with contaminant concentrations. The results of the RA reflect the accumulated variances of the individual measured values. A different kind of uncertainty stems from data gaps, i.e., additional information needed to complete the database for the assessment. Often, the data gap is significant, such as imprecision regarding the number of days that a youthful visitor might visit the Red Water Ponds Areas, or the absence of information on the effects of human exposure to a chemical (EPA, 1992c).

EPA (1992c) guidance urges risk assessors to address or provide descriptions of individual risk to include the "high end" portions and "central tendency" (CT) of the risk distribution. One way of fulfilling this request, if either cancer or noncancer risk exceed generally acceptable limits (cancer risk greater than 1E-5 or target organ-specific HI greater than 1), is to re-compute the ILCRs or HIs using CT values for as many intake model variables as possible. In contrast to the RME evaluation, which prevails in RAs, and uses upper-end values for intake or contact rates, exposure frequency and exposure duration, the CT evaluation chooses average or mid-range values for these variables (EPA, 1991). The intent is to present a quantified risk/hazard estimate more typical for the receptor of interest.

The CT exposure evaluation, however, falls short of its stated intent for several reasons. First, the same source-term concentration is usually used for the CT evaluation as is used for the RME evaluation. EPA (1993) considers that the UCL or MDC selected as a conservative estimate of average for the RME is appropriate for the CT estimates. Second, there is little information available as to what constitutes a reliable CT estimate for most exposure variables, with the possible exception of a simple on-site residential scenario. For these reasons, RME values are

still used. In addition, no CT toxicity values are available, so the uncertainty about the toxicity assessment is not included. A CT evaluation, therefore, usually provides little relief, compared with the RME, particularly for exposure scenarios such as the trespasser and construction worker, for which no reliable estimation of most exposure variable values can be made. It should be stated that management decisions are generally based on RME rather than CT evaluations. The CT evaluation, within the described limitations, simply adds perspective regarding the magnitude of uncertainty about the RME estimates.

Another method of quantifying uncertainty, called Monte Carlo simulation, provides a more graphic illustration of the uncertainty about a risk/hazard estimate, because it presents the risk as a range with probability densities. To be meaningful, however, Monte Carlo simulation requires that the nature of the distributions of the variables that drive the RA should be well characterized. However, well characterized distributions are available for few exposure or toxicological variables, in which case the Monte Carlo simulation provides an incomplete illustration of the magnitude or the distribution of the uncertainty. Should ILCRs or HIs exceed acceptable levels, Monte Carlo simulation, using Crystal Ball for EXCEL, will be considered to portray the uncertainty about the estimates.

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APPENDIX A

RESPONSES TO TECHNICAL REVIEW COMMENTS

TECHNICAL MEMO

TO: Mike Spangberg

DATE: 16 September 1998

FROM: Paul F. Goetchius

SUBJECT: **RED WATER PONDS AREA DRAFT HUMAN HEALTH RISK ASSESSMENT (HHRA) WORK PLAN (WP): RESPONSE TO COMMENTS, REVISED.**

COMMENTS FROM LANNAE LONG THROUGH M. ZOCCOLA 9 JUNE 1998

1. As per 20 July conference call between USACE, OEPA and IT Corp., surface soil is defined as 0 to 1 ft bgs and subsurface soil is defined as 1 to 10 ft bgs.

2. All stakeholders agree that overburden or perched groundwater, which is transient, is not a potential source of potable water. No plausible pathways exist by which human receptors are exposed to perched groundwater, which is not evaluated in the HHRA. Please see response to OEPA comment 6.

3. Agreed.

4.A. Agreed; please see response to OEPA comment 3.
4.B. Agreed; please see response to OEPA comment 3.
4.C. The following will be inserted after the first sentence in the penultimate paragraph in Section 2.1.3: "The UTL is the concentration, with a probability of 0.95 (or a confidence of 95 percent), that will capture (or cover) 95 percent of the background concentration. If the site data truly reflect background, there is a 5 percent probability that any site sample concentration will exceed the UTL."
4.D. The last sentence in the penultimate paragraph in Section 2.1.3 will be revised as follows: "The Wilcoxon Rank Sum test (STATISTICA™) is used for this purpose, because it is applicable to all data distributions."
4.E. Please see response to 4.D.

5. Please see response to OEPA comment 3.

6.A. Agreed; the receptor descriptions in Table 3-1 will be revised to provide the rationale for considering certain pathways as incomplete.
6.B. The fact that contact with bedrock groundwater is an incomplete exposure pathway under the current scenario is more defensible justification for not quantifying inhalation of VOCs than is the rationale offered by the reviewer; i.e., the large dilution factor of ambient air. The rationale offered by the reviewer is the more defensible justification for the future scenario,

which is consistent with the presentation in Figure 3-1.

6.C. Evaluation of inhalation of VOCs from soil for the construction worker scenario is a plausible complete exposure pathway ordinarily evaluated in baseline risk assessments, because excavation and grading may uncover subsurface sources or pockets of VOCs.

6.D. The pathway is correct as written.

COMMENTS FROM OEPA 22 JUNE 1998

1. As per 20 July conference call between USACE, OEPA and IT Corp., surface soil is defined as 0 to 1 ft bgs and subsurface soil is defined as 1 to 10 ft bgs.

2. USACE emphasized that DOD is only responsible to address contaminants release by DOD operations. In order to provide optimum information for risk managers, it was decided that the Risk Characterization section will present Total Site Risk (including all COPCs, even those present at natural or anthropogenic background or upgradient levels), Background Risk (restricted to those COPCs present at natural and anthropogenic background or upgradient levels), and Site-Related Risk (restricted to site-related COPCs; i.e., those for which background data are not available and those that exceed natural or anthropogenic or upgradient levels).

3. The following will be inserted before the last paragraph in Section 2.1.3: "The UTL is calculated as follows:

$$UTL = \overline{X} + k(a) \qquad \text{Eq. 2.1}$$

where:

UTL = upper tolerance limit (confidence factor of 0.95 and coverage of 95 percent)
$\bar{x}$ = arithmetic mean
a = standard deviation
k = tolerance factor (Table 5, OEPA [1991], Final ***How Clean is Clean Policy***, 26 July."

The last sentence in the last paragraph in Section 2.1.3 will be revised as follows: "Therefore it may be necessary to express the concentration of each metal in each sample as a percent of the total metal content of the sample before performing the comparisons."

4. Agreed.

5. The discussion of nonparametric UCL in Section 2.2.1 will be revised as follows: "The nonparametric UCL is estimated as the 95 percent UCL rank order on the arithmetic mean of the data set. It is estimated by ranking the data observations from smallest to largest. The arithmetic mean is converted to a percentile by interpolation. The rank order of the data point selected as the UCL is estimated from the following equation (Gilbert, 1987):

$$u = p(n+1) + Z_{1-\alpha}\sqrt{np(1-p)}$$

where:

u = rank order of value selected as upper confidence limit, calculated
p = percentile corresponding to the arithmetic mean
n = number of samples in the data set
α = confidence limit (95 percent)
$Z_{1-\alpha}$ = normal deviate variable."

6. During further discussion it was decided to estimate groundwater concentrations as described in the draft HHRA WP.

   During the discussion it became apparent that there is no plausible pathway by which current or future potential receptors would be exposed to shallow groundwater, which is not to be evaluated in the baseline risk assessment (BLRA). Potential exposure to bedrock (deep) groundwater involves its development as a source of potable water. However, analytical data for the bedrock groundwater evaluation are inadequate for use in risk assessment. A temporary solution is to postpone bedrock groundwater evaluation to a subsequent task, but this would result in inadequate evaluation of site exposure because it would not be possible to sum risks and hazards across media. Further deliberation by USACE, OEPA and IT Corp. concluded with the decision to include the protocol for groundwater analysis in the HHRA WP, but to acknowledge that data for bedrock groundwater evaluation are inadequate, and that risk/hazard estimates cannot be completed across media until adequate bedrock groundwater data are obtained, evaluated and used in the BLRA.

7. The EPA Soil Screening Level Technical Background Document has performed a sensitivity analysis on the leaching model and determined that site size has little impact on the result. This information will be added to the document.

8. The fence and patrols around the facility prevent predictable and repeated access to the site; therefore, the youthful visitor is not evaluated under the current exposure scenario. In the future, it is assumed that the site is released for unrestricted (possibly residential) development. Residential exposure to surface water and sediment would be more frequent than visitor exposure and would represent the upper-bound; therefore, the youthful visitor is not evaluated under the future exposure scenario. Pathways for exposure to sediment and surface water quantified in the RA are limited to incidental ingestion and dermal contact. Groundskeeper exposure to surface water and sediment, if it occurs at all, would be accidental. (Sediment is defined as soil covered by surface water.) Infrequent accidental exposure does not lend itself to analysis by the chronic toxicity risk assessment paradigm used in a baseline RA.

   The groundskeeper, resident and construction worker will be evaluated for incidental ingestion of and dermal contact with soil, and inhalation of dust and VOCs from soil.

9. The youthful visitor (trespasser) scenario will not be evaluated; please see response to comment 8.

The assumptions for residential exposure to soil will be modified to incorporate exposure to surface water and sediment. Care will be taken to ensure that exposures are not double-counted; i.e., contact with surface water and sediment precludes simultaneous exposure to soil. A fractional term will be developed to apportion exposure between the media. The fraction is justified, because incidental ingestion of soil and sediment is not a discrete event or incident-based. It is a continuous phenomenon arising largely from hand-to-mouth activity and inhalation of dust followed by ciliary clearance of the large airways and swallowing the residue.

Further discussion concluded with adoption of the following AF factors:

AF = 0.009 mg/cm^2 for dermal uptake from soil for the groundskeeper.
AF = 0.08 mg/cm^2 for dermal uptake from soil for the construction worker.
AF = 0.2 mg/cm^2 for dermal uptake from soil for the resident.
AF = 0.08 mg/cm^2 for dermal uptake from sediment for the construction worker.
AF = 0.2 mg/cm^2 for dermal uptake from sediment for the resident.

10. Please see discussion regarding AF in response to Comment 9.

11. Volatilization of VOCs from soil will be quantified for the groundskeeper, construction worker and on-site resident, as per subsequent discussions.

12. Groundskeeper exposure to VOCs in air from groundwater during a shower will not be quantified because the resident would reflect the upper bound on this pathway.

13. VOCs from surface water would be subject to the huge dilution factor of ambient air, reducing airborne concentrations to levels expected to be toxicologically insignificant; therefore, this pathway will not be quantified for the construction worker.

The AF for the construction worker will be revised as described in response to Comment 9.

14. Please see response to comment 11.

The resident will be evaluated for exposure to surface water and sediment; please see response to comment 8.

15. Agreed; a whole-house model will be substituted for the shower model to estimate average concentrations in indoor air to which both the adult and child resident would be exposed. Section 3.2.1.3 will be revised as follows:

"*3.2.1.3 VOC Concentrations from Groundwater*

Inhalation of VOCs released from groundwater used as household water is evaluated for the resident. A model for the rate of VOC release from groundwater is used to estimate an average concentration of VOCs in air, as follows:

$$C_a = \frac{W_{house} \cdot K_m \cdot C_g \cdot t}{VR_{house} \cdot T}$$

where:

C_a	=	concentration of VOC in indoor air, average for entire house adjusted to reflect average time spent indoors (mg/m^3, calculated).
W_{house}	=	average daily household indoor water use (447 L/day, see below).
K_m	=	VOC mass transfer coefficient, all household uses, (0.54, unitless, see below)
C_g	=	concentration of VOC in groundwater (mg/L)
t	=	time spent by resident indoors (16.4 hours/day, EPA, 1996c)
VR_{house}	=	average daily ventilation rate (3984 m^3/day, see below).

EPA (1996c) reviewed several studies of the volumes of houses and air exchange rates, and recommends 369 m^3 as a central estimate of the volume of a house, and 0.45 air changes/hour as a typical air exchange rate. A ventilation rate for a typical house of 166 m^3/hour (3984 m^3/day) is estimated from these values. EPA (1996a) also related residential volumes to household size. The average volume of a 1-person household is 269 m^3; the average volume of a 2-person household is 386 m^3. The average volume of all households, 369 m^3 as described above, is similar to the volume for a 2-person household. Therefore, it is assumed that the average household contains two persons. EPA (1996a) estimated a mean per capita indoor water use rate of 59 gallons/day. Assuming a 2-person household, this estimate is equivalent to 118 gallons/day, or 447 L/day.

McKone and Knezovich (1991) measured the transfer of trichloroethene (as a typical VOC) from shower water to shower room air and obtained an average mass transfer coefficient of 0.6. They noted that this was similar to the value for radon (0.70 for showers and 0.54 for all household uses of water). They also observed that when the dimensionless Henry's law constant is greater than 1E-1, mass transfer is almost completely dependent on the diffusion coefficient of the chemical in water; however, this parameter varies little for most organic chemicals in the 100 to 500 g/mole range. Therefore, the mass transfer coefficient for radon of 0.54 estimated for all household uses of water is used for all VOCs in this evaluation."

Additional reference: McKone, T.E. and J.P. Knezovich, 1991, "The Transfer of Trichloroethylene (TCE) from a Shower to Indoor Air: Experimental Measurements and

EPA (1996a) estimated a mean per capita indoor water use rate of 59 gallons/day. Assuming a 2-person household, this estimate is equivalent to 118 gallons/day, or 447 L/day.

McKone and Knezovich (1991) measured the transfer of trichloroethene (as a typical VOC) from shower water to shower room air and obtained an average mass transfer coefficient of 0.6. They noted that this was similar to the value for radon (0.70 for showers and 0.54 for all household uses of water). They also observed that when the dimensionless Henry's law constant is greater than 1E-1, mass transfer is almost completely dependent on the diffusion coefficient of the chemical in water; however, this parameter varies little for most organic chemicals in the 100 to 500 g/mole range. Therefore, the mass transfer coefficient for radon of 0.54 estimated for all household uses of water is used for all VOCs in this evaluation."

Additional reference: McKone, T.E. and J.P. Knezovich, 1991, "The Transfer of Trichloroethylene (TCE) from a Shower to Indoor Air: Experimental Measurements and Their Implications," ***Journal of the Air and Waste Management Association***, 41: 282-286.

Section 3.2.2 will be revised as follows to reflect the changes described in Section 3.2.1.3:

3.2.2 Exposure-Point Concentrations of VOCs in Groundwater: Residential Dermal Uptake

As noted above, volatilization of VOCs from household water reduces the concentration remaining available for dermal contact (it is assumed that groundwater used as drinking water is ingested before significant volatilization occurs). The concentration of VOCs remaining in the water are calculated by difference as follows:

$$C_d = C_g \cdot (1 - K_m)$$

where:

C_d	=	concentration of VOC in household water available for dermal exposure (mg/L, calculated)
C_g	=	concentration of VOC in groundwater (mg/L)
K_m	=	VOC mass transfer coefficient, all household uses, (0.54, unitless)."

The AF for the residential receptor will be 0.2 mg/cm^2.

Their Implications," ***Journal of the Air and Waste Management Association***, 41: 282-286.

Section 3.2.2 will be revised as follows to reflect the changes described in Section 3.2.1.3:

3.2.2 Exposure-Point Concentrations of VOCs in Groundwater: Residential Dermal Uptake

As noted above, volatilization of VOCs from household water reduces the concentration remaining available for dermal contact (it is assumed that groundwater used as drinking water is ingested before significant volatilization occurs). The concentration of VOCs remaining in the water are calculated by difference as follows:

$$C_d = C_g \cdot (1 - K_m)$$

where:

C_d	=	concentration of VOC in household water available for dermal exposure (mg/L, calculated)
C_g	=	concentration of VOC in groundwater (mg/L)
K_m	=	VOC mass transfer coefficient, all household uses, (0.54, unitless)."

The AF for the residential receptor will be 0.2 mg/cm^2.

16. VOCs from surface water would be subject to the huge dilution factor of ambient air, reducing airborne concentrations to levels expected to be toxicologically insignificant; therefore, this pathway will not be quantified for the resident.

17. The default ABS values provided by OEPA will be used where empirical data are not sufficient to refine the estimate.

18. Agreed.

19. Agreed. It is expected that chemicals of concern, if any, will be limited to a few nitroaromatics. Care will be taken to ensure that chemicals remaining at their RBRCs will not yield unacceptable cumulative risk or hazard.

Ecological Risk Assessment Work Plan
Red Water Ponds Areas
Plum Brook Ordnance Works
Sandusky, Ohio

Prepared for:

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Revision 1

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List of Acronyms

ARP	assessment receptor profile
BAF	bioaccumulation factor
BCF	bioconcentration factor
BEA	baseline ecological risk assessment
BTAG	Biological Technical Assistance Group
COPEC	chemical(s) of potential ecological concern
DOD	U.S. Department of Defense
EPA	U.S. Environmental Protection Agency
ERA	ecological risk assessment
ESCM	ecological site conceptual model
ET	Ecotox Threshold
FCM	food chain multiplier
HQ	hazard quotient
IT	IT Corporation
K_{oc}	soil adsorption coefficient
K_{ow}	octanol-water partition coefficient
LOAEL	low observable adverse effects level
MDC	maximum detectable concentration
NOAEL	no observable adverse effects level
ODNR	Ohio Department of Natural Resources
OME	Ontario Ministry of the Environment and Energy
PBOW	Plum Brook Ordnance Works
PCB	polychlorinated biphenyl
QC	quality control
RBSEV	risk-based screening ecotoxicity value
RTE	rare, threatened, or endangered
RTV	reference toxicity value
SQL	sample quantitation limit
UCL	upper confidence limit
UTL	upper tolerance limit

1.0 Introduction

A baseline ecological risk assessment (BEA) will be performed to provide an estimate of current and future ecological risk associated with potential hazardous substance releases within the Red Water Ponds Areas at the former Plum Brook Ordnance Works (PBOW) in Sandusky, Ohio. Ecological risks will be assessed separately for logically distinct source areas within the facility. The results of the BEA will contribute to the overall characterization of the sites and serve as part of the baseline used to develop, evaluate, and select appropriate remedial alternatives. The ecological risk assessment will be performed following the general guidelines of the *Tri-Service Procedural Guidelines for Ecological Risk Assessments* (Wentsel, et al., 1996), with secondary guidance from *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (U.S. Environmental Protection Agency [EPA], 1997), and *Region 5 Biotechnical Assistance Group (BTAG) Ecological Risk Assessment Guidance Bulletin No. 1* (EPA, 1996a).

The BEA will evaluate the West Area Red Water Ponds and the Pentolite Road Red Water Ponds Areas. The primary objective of the BEA is to determine whether unacceptable adverse risks are posed to ecological receptors as a result of potential hazardous substance releases. This objective is met by characterizing the ecological communities in the vicinity of the subsites, determining the particular hazardous substances being released from the subsites, identifying pathways for receptor exposure, and estimating the magnitude of the likelihood of potential risk to identified receptors. The BEA will address the potential for adverse effects to the vegetation, wildlife, aquatic life (including both fish and aquatic macro-invertebrates), endangered and threatened species, and wetlands or other sensitive habitats associated with the subsites.

Concentrations of chemicals will be measured in relevant environmental media including soil and groundwater. Using this information, concentration data will be used by IT Corporation (IT) to perform a Tier 1 BEA, including a problem formulation (Chapter 2.0); an exposure characterization (Chapter 3.0); an ecological effects characterization (Chapter 4.0); and a risk characterization (Chapter 5.0). These subtasks are described in more detail below.

IT will evaluate 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. 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 the need for a predictive assessment is needed. The criteria by which a predictive assessment is needed 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:

- There is no 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.
- There is no potential for adverse ecological effects to ecological entities at the site is minimal or nonexistent due to the lack of **potential ecological receptors**.
- There is no potential for adverse ecological effects to ecological entities at the site is minimal or nonexistent due to the lack of **potential exposure pathways**.
- There is no potential for adverse ecological effects to ecological entities at the site is minimal or nonexistent due to the lack of **potential chemical stressors**.

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

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

2.1 Ecological Site Description

IT will describe the site in sufficient detail to ensure that the CELRN technical specialist can be oriented to the site. This information will be assembled from existing sources, without conducting additional field studies. IT will contact natural resource personnel (e.g., Federal or State Officials) to obtain any relevant data or useful ecological information.

2.2 Pre-Assessment Reconnaissance (Biota Checklist)

IT will perform a site reconnaissance. The primary objective of this reconnaissance will be to collect qualitative information on the type, quality, and location of biological resources at the subsites. This will be achieved as follows:

- Dominant plant species will be identified, and plant communities will be defined based on dominant species observed.
- Observations of fauna will be made. Mammals will be identified by tracks, scat, burrows, and sightings. Bird, reptile, and amphibian identifications will be made by sightings. Fish and aquatic macroinvertebrates will be collected for identification as necessary, depending on characteristics of each subsite.
- Areas will be examined for vegetative stress. Stress may be exhibited by stunted growth, poor foliage growth, tissue discoloration, and a loss of leaf coverage. Due to the seasonal component of this evaluation, the survey will be performed during late spring or early summer, as the schedule permits.

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

The site reconnaissance will be performed by two IT ecologists. Prior to arrival at the site, IT personnel will obtain relevant information on the site, 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 site. Additionally, the *Biological Inventory of Plum Brook Station, 1994* (National Aeronautics and Space Administration, 1995) which identifies and shows the locations of threatened and endangered species at PBOW, will be reviewed. IT personnel will complete a checklist similar to that on EPA's "Checklist for Ecological Assessment/Sampling" (EPA, 1997); in situ water column measurements (i.e., pH, temperature, dissolved oxygen, conductivity) will not be collected. The location of known or potential contaminant sources affecting the site and the probable gradient of the pathway by which contaminants may be released from the site to the surrounding environment will be identified. IT personnel will use the reconnaissance to evaluate the site for more subtle clues of potential effects from contaminant release. IT will determine the designation of Pipe Creek and any other waters potentially impacted by contaminant migration.

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

The methods used to characterize natural resources will focus on aquatic and terrestrial resources at the subsites and within the immediate vicinity. If not already in existence, general habitat maps will be prepared showing the type and extent of biological communities present within the immediate vicinity of each subsite. 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

IT will determine if the site has 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' (ODNR) Division of Natural Areas and Preserves. The site reconnaissance will not include wetlands delineation activities.

2.4 Significant Ecological Threats

IT will determine whether significant ecological threats exists and whether these threats are related to chemical contamination caused by Department of Defense (DOD) activities. The initial screening of whether significant threats exist will be based on the qualitative absence of biota or animal life in areas expected to support these ecological components.

2.5 Review, Evaluation, and Presentation of Analytical Data

IT will review and evaluate analytical data collected by Dames & Moore during 1994, as well as all previous and ongoing IT investigations. Data identified as being of acceptable quality for use

in the BEA will be summarized in a manner that presents the pertinent information to be applied in the BEA. Any data rejected during the data evaluation as a result of the data evaluation will be identified, along with the rejection rationale.

2.6 Selection of Preliminary Chemicals of Ecological Potential Concern

IT will identify a subset of chemicals detected at the site that have data of good quality and are not naturally occurring or a result of nonsite sources. The chemicals must also be present at sufficient frequency, concentration, and location to pose a potential risk to ecological receptors. Examples of screening criteria that will be used include the following: analytical detection limit; frequency of detection less than 5 percent; comparability with background; role as an ecologically essential nutrient at site concentrations; and comparability with ecologically relevant screening criteria. This selection process is described in more detail in the following subsections.

2.6.1 Data Organization

The data for each chemical will be sorted by medium. For ecological impacts, soil from 0 to 6 feet will be considered. Chemicals which are not detected at least once in a medium will not be included in the risk assessment. Available background data will be determined for each medium. Potential sources of background information will include data from previous and current investigations, as well as monitoring wells in areas unaffected by site activities.

The analytical data may have qualifiers from the analytical laboratory quality control (QC) 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 (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 - QC 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 five-times, ten-times 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 later in 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. In general, "outliers" are included in the calculation of the UCL because high values in environmental data are seldom true statistical outliers. Inclusion of outliers increases the overall conservatism of the risk estimate, and the likelihood of rejecting the null hypothesis.

Data sets will be tested for normality and lognormality based on the Shapiro-Wilks test (EPA, 1992a). Statistical analysis will be performed only on those chemicals whose maximum detected concentration (MDC) exceeds the risk-based screening ecotoxicity values (RBSEV). If statistical tests support the assumption that the data set is normally distributed, the UCL for a normal distribution is calculated. If the statistical analysis shows the data to be lognormally distributed, the UCL is calculated for a lognormal distribution.

The UCL is calculated for a normal distribution as follows (EPA, 1992b):

$$UCL = \bar{x} + t_{1-\alpha, n-1} \; x \; (s/\sqrt{n}) \qquad \text{Eq. 1}$$

where:

$\bar{x}$ = sample arithmetic mean
t_1 = critical value for student's plus distribution
α = 0.05 (95 percent confidence limit for a one-tailed test)
n = number of samples in the set
s = sample standard deviation.

The UCL is calculated for a lognormal distribution as follows (Gilbert, 1987):

$$UCL = e^{\left(\bar{y} + (0.5 \cdot s_y^2) + \left[H_{0.95} \cdot \frac{s_y}{(n-1)^{0.5}}\right]\right)} \qquad \text{Eq. 2}$$

where:

$\bar{y}$ = $\sum y/n$=sample arithmetic mean of the log-transformed data, y = ln x
s_y = sample standard deviation of the log-transformed data
n = number of samples in the data set

$H_{0.95}$ = value for computing the one-sided 95 percent UCL on a lognormal mean from standard statistical tables (Land, 1975).

A nonparametric confidence limit is used when the data set fits neither a normal or a lognormal distribution. The nonparametric UCL is estimated as the 95 percent UCL rank order on the arithmetic mean of the data set. It is estimated by ranking the data observations from smallest to largest. The arithmetic mean is converted to a percentile by interpolation. The rank order of the observation selected as the UCL is estimated from the following equation (Gilbert, 1987):

$$u = p(n + 1) + Z_{1-\alpha}\sqrt{np(1 - p)} \qquad \text{Eq. 3}$$

where:

u = upper confidence limit
p = percentile corresponding to the arithmetic mean
n = number of samples in the set
α = confidence limit; 95 percent
$Z_{1-\alpha}$ = normal deviate variable.

Analytical results are presented as "nondetects" ("U" qualifier) whenever chemical concentrations in samples do not exceed the detection or quantitation limits for the analytical procedures for those samples. Generally, the detection limit is the lowest concentration of a chemical that can be "seen" above the normal, random noise of an analytical instrument or analytical method. To apply the previously mentioned statistical procedures to a data set with nondetects, a concentration value must be assigned to nondetects. Nondetects are assumed to be present at one-half the sample quantitation limit (SQL), although judgement is used in those cases where matrix interference or other phenomena drive the SQL unusually high. The UCL or the MDC, whichever is smaller, is selected as the source-term concentration, and is understood to represent a conservative estimate of average for use in the risk assessment or in various transport models used to estimate exposure-point concentrations.

2.6.3 Frequency of Detection

Chemicals that are detected infrequently may be artifacts in the data that may not reflect site-related activity or disposal practices. These chemicals will not be included in the risk evaluation. Generally, chemicals that are detected only at low concentrations in less than 5 percent of the samples from a given medium are dropped from further consideration, unless their presence is expected based on historical information about the site. 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 Natural Site Constituents (Background and Essential Nutrients)

Chemical concentrations will be compared to background concentrations 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 judgement in resolving site-related issues for metals, since metals are naturally present in most environmental media. The statistical techniques generally involve comparing the site data with background data.

The first statistical technique is the development of an upper tolerance limit (UTL) for background, and comparing the MDC with the UTL. Chemicals with MDCs less than the background UTL are eliminated from further consideration. If the MDC exceeds the UTL, the chemical is retained as a COPEC, or a more rigorous statistical analysis may be performed. The statistical analysis consists of comparing the site and background data sets to determine if both are drawn from the same population. The Wilcoxon Rank Sum test is used for this purpose.

It must be understood that statistical analysis is only a tool to aid the exercise of professional judgement. Site data from uncontaminated areas with concentrations at the high end of background may "fail" statistical testing because of the limitations of sample size, i.e., the full range of actual background and site variation was not captured. Statistical testing is based on absolute values, but the approximately 20 metals generally analyzed together constitute only approximately 4 to 5 percent of a given sample. Apparently high values of one or more metals may arise from a diminished amount of other constituents in soil, e.g., silica or organic matter, that may be more abundant in background areas. Therefore, it may be necessary to normalize the metal concentrations in site and background data before performing comparisons.

Essential nutrients such as calcium, chloride, iodine, magnesium, phosphorus, potassium and sodium may be eliminated as COPEC, provided that their presence in a particular medium is judged to be unlikely to cause adverse effects on wildlife.

2.6.5 Comparison to Risk-Based Screening Ecotoxicity Values

A comparison will be made between MDCs of chemicals in media and RBSEV for ecological endpoints following recommendations in EPA Region 5 BTAG Bulletin No. 1 (EPA, 1996a). Chemicals that exceed the RBSEVs will be retained as COPEC. The following RBSEV will be used for the ecological evaluation:

- ***Soil.*** Soil screening values from EPA Region 3 BTAG screening levels (EPA, 1995a), and other available guidance documents, as appropriate.
- ***Groundwater.*** If groundwater is known to impact surface water at the site, EPA Ecotox Threshold (ET) screening values for freshwater (EPA, 1996b) and Ohio EPA Water Quality Criteria will be used.
- ***Surface Water.*** EPA ET screening values for freshwater (EPA, 1996b) and Ohio EPA Water Quality Criteria will be used.
- ***Sediment.*** EPA ET values for freshwater sediment and sediment criteria from the Ontario Ministry of the Environment and Energy (OME) (OME, 1993) will be used, as recommended in EPA Region 5 guidance (EPA, 1996a).

2.6.6 Summary of COPEC Selection

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

- Chemical name
- Frequency of detection
- Range of detected concentrations
- Range of detection limits
- Arithmetic mean (average) of site concentrations
- Distribution type
- UCL of the mean of the concentration
- Source-term concentration
- Appropriate RBSEV
- Twice the value of the arithmetic mean of background concentrations of inorganic chemicals

- 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 COPEC due to other considerations. Examples of these exceptions include: potential break-down products, chemicals with detection limits greater than the RBSEV, 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 COPEC and will be quantitatively evaluated in the BEA. The physical, chemical, and toxicological properties of the identified COPEC will be reviewed from the scientific literature and summarized in COPEC profiles. When possible, data and information directly relevant to the BEA will be included in the COPEC profiles. COPEC-specific information pertaining to physiological, biological, or ecological effects that is used directly in the exposure and effects analysis of this BEA may be presented and discussed in the COPEC profiles. In addition, justification for the use of surrogate chemical data in the absence of direct chemical data for COPEC may be presented and discussed in the profiles. The COPEC profiles will be included in the final ecological risk assessment (ERA) report as an appendix.

2.7 Selection of Assessment Receptors

IT will select assessment receptors for evaluation during this BEA. In order to focus the exposure characterization portion of the BEA 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, IT will focus the selection process on species, groups of species, or functional groups, rather than higher organization levels such as communities or ecosystems. Site biota will be organized into major functional groups. For terrestrial communities, the major groups are plants and wildlife, including terrestrial invertebrates, mammals, and birds. For aquatic and/or wetland communities, the major groups are flora and fauna, including vertebrates (water fowl and fish), aquatic invertebrates, and wetland/terrestrial mammals. Species presence and relative abundance will be determined during the site reconnaissance prior to identification of target species.

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

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

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

Results of the assessment receptor selection process will be presented in detailed biological and ecological descriptions called assessment receptor profiles (ARP). Additionally, the biologically relevant criteria used to select it as an assessment receptor will be discussed and summarized in the ARP. The ARPs will be included in the final ERA report as an appendix.

2.8 Ecological Endpoint (Assessment and Measurement) Identification

The protection of ecological resources, such as habitats and species of plants and animals, is a principal motivation for conducting the BEA. Potential ecological assessment and measurement endpoints will be proposed after the site reconnaissance and a thorough review of existing reports

and site-related documents. The final assessment and measurement endpoints will be selected by agreement between risk assessors, risk managers, and regulatory agencies.

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

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

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

Measurement endpoints for this Tier I BEA 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. COPEC 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

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

3.0 Exposure Characterization

IT will develop an estimate of the nature, extent, and magnitude of potential exposure of assessment receptors to COPEC that are present at or migrating from the site, considering both current and reasonably plausible future use of the site. Exposure characterization is critical in further evaluating the risk of compounds identified as COPEC during the selection 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 areas may contact contaminants.

3.1 Exposure Analysis

IT will perform an exposure analysis, which will combine the spatial and temporal distribution of the ecological receptors with those of the COPEC 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.

Calculation of plant uptake values is not necessary as the 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 COPEC found in surface water, surface soil, and sediment. Exposure rates for terrestrial wildlife receptors will be based solely upon ingestion of contaminants from these media and from consumption of other organisms. Given the scarcity of data available 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 exception of burrowing animals and dust-bathing birds.

The first step in estimating exposure rates for terrestrial wildlife involves the calculation of feeding and watering 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 watering rates, and dietary composition are available for many species, or may be estimated using allometric equations (Nagy, 1987). Data will be gathered on incidental ingestion

of soil, and will be incorporated for the receptor species. This information will be summarized and documented in the ARPs.

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

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

For aquatic faunal receptors, the calculation of exposure rates will depend on the determination of the contaminant concentration in water and on food-chain multipliers, bioconcentration factors (BCF), and bioaccumulation factors (BAF). If appropriate, an evaluation will be made of the time each organism spends associated with surface water or sediment pore water in order to modify exposure rates.

For species exposed to organic contaminants found in sediment, calculations have been performed to quantify interstitial (pore) water contaminant concentrations given a known sediment concentration. Suter (1993) notes an algorithm to calculate pore water concentrations for nonionic organic chemicals, as follows:

$$\text{Pore water concentration (milligrams per liter)} = (SC)/(F_{oc})\,(K_{oc}) \qquad \text{Eq. 4}$$

where:

SC = sediment concentration (milligram per kilogram)
F_{oc} = fraction organic carbon in sediment (kg organic carbon/kg sediment)
K_{oc} = chemical-specific organic carbon partition coefficient (L/kg).

Ecological routes of exposure for biota may be direct (bioconcentration) or through the food web via the consumption of contaminated organisms (biomagnification). Direct exposure routes include dermal contact, absorption, inhalation, and ingestion. Examples of direct exposure

include animals incidentally ingesting contaminated soil or sediment (e.g., during burrowing or dust-bathing activities); animals ingesting surface water; plants absorbing contaminants by uptake from contaminated sediment or soil; and the dermal contact of aquatic organisms with contaminated surface water or sediment.

Food web exposure can occur when terrestrial or aquatic fauna consume contaminated biota. Examples of food web exposure include animals at higher trophic levels consuming plants or animals that bioaccumulate contaminants. The concepts of bioconcentration, bioaccumulation, and biomagnification are used throughout this document. Definitions describing their application are presented in the Glossary of Terms (Appendix A).

Contamination of biota could result from exposure to one or more COPEC. 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.

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 BEA. If any of these components are 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.

Adjustments will be made for potential biomagnification of contaminants through aquatic trophic levels. Food chain multipliers (FCM), derived by EPA (1995b), will be used to assess the possibility of contaminant magnification through site receptors. The FCMs are multiplied by chemical-specific BCFs to obtain BAFs. The BEA will either use laboratory-measured BCF values obtained from the scientific literature or fish BCFs will be calculated for organic compounds using the following equation (EPA, 1995b):

$$BCF = K_{ow} \qquad \text{Eq. 5}$$

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$$BCF = K_{ow} \qquad \text{Eq. 5}$$

where:

K_{ow} = chemical-specific octanol/water partition coefficient.

When possible, K_{ow} values for appropriate COPEC will be obtained from the literature or from databases, and will be listed among the fate and transport properties within the COPEC profiles.

The BCF is dependent upon a chemical-specific K_{ow} that relates to a chemical's tendency to partition to a polar versus nonpolar solution. EPA has established a relationship between the K_{ow} and the FCM such that as the K_{ow} increases, the FCM increases correspondingly.

For sediment or soil, the percent carbon present is critical to partitioning. For these matrices, the K_{ow} will be converted to a K_{oc} value (EPA, 1996c) as follows:

$$\log K_{oc} = 0.00028 + (0.983 \times \log K_{ow}) \qquad \text{Eq. 6}$$

where:

K_{oc} = the partition constant relative to organic carbon.

This equation was chosen because it is the best fit for site-related compounds, semivolatile nonionizing organic compounds.

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

- A measured BAF for an inorganic or organic chemical derived from a field study
- A predicted BAF for an organic chemical derived from a field-measured biota-sediment accumulation factor
- A predicted BAF for an inorganic or organic chemical derived from a laboratory-measured BCF and a FCM
- A predicted BAF for an organic chemical derived from a K_{ow} and a FCM.

The EPA guidance notes, however, that for chemicals for which no K_{ow} is available, and for which no BCF is calculable, a default FCM of 1.0 should be used. Thus, for inorganics not

thought to biomagnify and/or which no literature value is available, this value of 1.0 will be used at each trophic level.

In addition to the aquatic food web, FCMs are also related to an organism's trophic status as predator/prey, producer/consumer, etc in the terrestrial food web. Although exposures of terrestrial floral and faunal receptors are significant considerations for many hazardous waste sites, well accepted models for predicting the fate of many contaminants in terrestrial systems are less developed. Trophic level compartments and transfer between compartments based on uptake, storage, and loss processes are not as well defined in terrestrial systems as in aquatic systems. In addition, the relationship between K_{ow} and bioconcentration is less well delineated by trophic level in terrestrial ecosystems. For the current BEA, soil-to-plant and food-to-muscle BAFs will be estimated for organic constituents using the log K_{ow} relationships developed by Travis and Arms (1988). Soil-to-insect BAFs will be based on log K_{ow} relationships developed by Connell and Markwell (1990). Inorganic constituent BAFs will be based on literature values such as those found in Baes, et al. (1984), International Atomic Energy Agency (1994), and Ma (1982).

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

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

For plant exposure, soil samples taken from 0 to 6 feet (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 whether they can be chemically mobilized (in a bioavailable form) and released for plant absorption. Generally, neutral to alkaline soils (soil pH of 6.5 or greater) restrict the absorption of toxic metals, making pathway completion to plants difficult. Literature values for soil-to-plant transfer rates for inorganic and organic soil contaminants and for organic soil contaminants will be used unless contaminant-specific information is available.

Sediment Exposure Pathway. Sediment consists of materials precipitated or settled out of suspension in surface water. Potential contaminant sources for sediment include buried or stored waste, and contaminated surface water, groundwater, and soil. The release mechanisms include surface water runoff, groundwater discharge, and airborne deposition. Potential receptors of chemicals in contaminated sediment include aquatic flora and fauna. Direct exposure routes for contaminated sediment include uptake by aquatic flora and ingestion by aquatic fauna. Indirect exposure pathways from sediment include consumption of bioaccumulated contaminants by consumers in the food chain. Chemical bioavailability of many nonpolar organic compounds (e.g., polychlorinated biphenyls [PCB] and pesticides) decreases with increasing concentrations of total organic carbon in the sediment; however, these compounds can still bioaccumulate up the food chain (Landrum and Robbins, 1990).

Surface Water Exposure Pathway. Surface water represents a potential transport medium for COPEC. Potential sources for contaminated surface water include: buried or stored waste, stored or spilled fuel, contaminated soil and groundwater, and deposition of airborne contaminants. The release mechanisms include surface runoff, leaching, and groundwater seepage. Potential receptors of contaminated surface water include terrestrial and aquatic fauna and aquatic flora. Exposure routes for contaminated surface water include ingestion by terrestrial fauna, and uptake and absorption by aquatic flora and fauna. Consumption of bioaccumulated contaminants constitutes a potential indirect exposure pathway for faunal receptors. Chemical bioavailability of some metals and other chemicals is controlled by water hardness, pH, and total suspended solids.

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

Groundwater itself is not an exposure point. However, contaminant transport along the shallow groundwater pathway is considered an exposure route to aquatic life, wetlands, and some wildlife where the groundwater discharges to surface water. This pathway is of importance to aquatic and wetland receptors if groundwater is found to be discharging to surface water.

3.2 Exposure Characterization Summary

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

4.0 Ecological Effects Characterization

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

4.1 Selection of Literature Benchmark Values

IT will consult appropriate sources for literature benchmark values. The level of effort will be limited to documents that summarize the available ecotoxicological information and will not consist of a review of the primary toxicological literature (i.e., IT will not review details of toxicity test conditions to determine validity of the tests performed).

4.2 Development of Reference Toxicity Values

IT will develop or determine reference toxicity values (RTV) for the site. These RTVs 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 COPEC, data on surrogate species and/or on endpoints other than the no observable adverse effects level (NOAEL) and lowest observable adverse effects level (LOAEL) may have to be used. The NOAEL is a dose of each COPEC that will produce no known adverse effects in the test species. The NOAEL was judged to be an appropriate toxicological endpoint since it would provide the greatest degree of protection to the receptor species. In addition, the LOAEL will be used as a point of comparison for decision-making for risk management purposes. In addition, in instances where data are unavailable for a site-associated COPEC, toxicological information for surrogate chemicals may have to be used. Safety factors will be used to adjust for these differences and extrapolate risks to the sites' receptors at the NOAEL and/or LOAEL endpoint. This process is described in the following paragraphs.

Toxicity information pertinent to identified receptors will be gathered for those analytes identified as COPEC. 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 BEA.

Using the relevant toxicity information, RTVs will be calculated for each of the COPEC. RTVs represent NOAELs and LOAELs with safety factors incorporated for toxicity information derived from studies other than no-effects or lowest-effects studies, and studies on species other than the receptors selected for this risk assessment. RTVs will be calculated using safety factors specified in Wentsel, et al. (1996). Interclass toxicity extrapolations will not be performed as physiological differences between classes are too great to be addressed with the use of simplistic safety factors. Separate factors are recommended to account for extrapolation to the no effects or lowest-effects endpoints, for study duration, and for extrapolation across taxonomic groups (e.g., species, genus, family, order). Additional safety factors will be employed for endangered species, as appropriate. These factors are multiplied together to derive a total safety factor. The reported effects dose is then adjusted to account for potential uncertainties by dividing by the total safety factor. Because NOAELs for the selected wildlife receptor species will most likely be based on NOAELs from test species, the latter will be converted to NOAELs specific to the selected wildlife receptors using a power function of the ratio of body weights, as described by Sample, et al. (1996). A body weight scaling factor of 0.25 will be used for mammals, whereas a body weight scaling factor of 0 will be used for birds.

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

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

5.0 Risk Characterization

The risk characterization phase integrates information on exposure, exposure-effects relationships, and defined or presumed target populations. The result is a determination of the likelihood, severity, and characteristics of adverse effects to environmental stressors present at a site. A semiquantitative approach to estimate the likelihood of adverse effects occurring as a result of exposure of the selected site receptors to COPEC will be taken. RTVs 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 RTV. HQs are a means of estimating the potential for adverse effects to organisms of a contaminated site, and for assessing the potential that toxicological effects will occur among site receptors.

5.1 Risk Estimation

IT will estimate the risk associated with the site. The risk estimation will be performed through a series of quantitative HQ calculations that compare receptor-specific exposure values with RTVs. The HQs will be compared to HQ guidelines for assessing the risk posed from contaminants. HQs less than or equal to 1 present no probable risk; HQs from 1 up to, but less than 10, present a low potential for environmental effects; HQs from 10 up to, but less than 100, present a significant potential that effects could result from greater exposure; and HQs greater than 100 present the highest potential for expected effects (Wentsel, et al., 1996).

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., organochlorine pesticides, PAHs, and phthalates). The following criterion will be used to determine if HQs will be summed: for a given receptor, only HQs for those chemicals that have a similar mode of toxicological action will be summed. While individual contaminants may affect distinct target organs or systems within an organism, classes of chemicals may act in similar ways, thus being additive in effect.

5.2 Uncertainty Analysis

The results of the BEA will be influenced to some degree by variability and uncertainty. In theory, investigators might reduce variability by increasing sample size of the media 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 COPEC that continue to show a relatively high potential for ecological risk. Since 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).

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 uncertainty analysis will be presented in part as a table listing the assumptions made for the ERA; 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 (quantitative [percent difference], or qualitative [high, medium, low, or unknown]); and, if possible, a description of recommendations for minimizing the identified uncertainties if the ERA progresses to higher level assessment phases (EPA, 1992b). The uncertainty analysis will identify and, if possible, quantify the uncertainty in the individual preliminary scoping assessment, problem formulation, exposure and effects assessment, and risk characterization phases of this BEA.

5.3 Risk Description

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

6.0 Risk Summary and Identification of Preliminary Remedial Action Objectives

IT will summarize ecological risk associated with releases from the site. This summary will be supported by tasks performed during the previous sections. Additionally, IT will make recommendations for further risk investigation, if appropriate. Finally, IT will develop site-specific remedial action objectives for the site.

7.0 Conclusions and Recommendations

Only the data, results, and conclusions of the various preliminary scoping and predictive assessment phases will be described. No recommendations concerning types of remedial actions to be conducted will be given other than to present the specific remedial action objectives. Conclusion and recommendations derived from the risk assessment will be based on the responses to the assessment hypotheses. The predictive assessment results will be summarized and presented in table format. These tables may serve as the foci of discussions with risk managers and regulatory agencies concerning the potential need for additional assessment at PBOW to reduce the uncertainty in the estimate of ecological risk.

8.0 References

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APPENDIX A

GLOSSARY OF TERMS

Appendix A

Glossary of Terms

Bioconcentration. For aquatic organisms, bioconcentration is the uptake and retention of a substance by an aquatic organism from the surrounding water through gill membranes or other external body surfaces. Terrestrial bioconcentration focuses on uptake and retention of contaminants from the surrounding medium on the organism level (as by the earthworm, for example).

Bioaccumulation. This refers to the uptake and retention of a substance by an aquatic organism from its surrounding medium and food (EPA, 1995b). Terrestrial bioaccumulation, as with aquatic bioaccumulation, is defined as an organism's uptake and retention of a substance from its surrounding medium and food.

Biomagnification. This refers to the process by which tissue concentrations of bioaccumulated toxic substances increase as the substances pass up through two or more trophic levels. The definition of this term is similar for both terrestrial and aquatic organisms.

APPENDIX B

RESPONSE TO TECHNICAL REVIEW COMMENTS

Appendix B

Plumbrook Ordnance Works
Red Water Pond Area Work Plan
Response to USCOE and OEPA Ecological RA Comments

Nashville District Corps of Engineers Responses

Comment No.	Response
7	The text will be modified to state that terrestrial and aquatic ecological conceptual site models will be developed, as appropriate.
8	It will be stated that chemicals which are not detected at least once will not be included in the risk assessment.
9	The misspelling will be corrected.
10a	It will be clearly stated the arithmetic mean is defined as the average.
10b	Data distribution type will be included as a field.

OEPA Responses

Comment No.	Response
B1	The size and quality of habitat deemed non-viable or absent of potential receptors will be fully described if applicable.
B2	See response to comment A5.
B3	See response to comment A3.
B4	Societal relevance will not be used as a selection criteria. The reference to societal relevance will be deleted.
B5	This sentence will be deleted.
B6	Disagree. IT takes the position that the Bintein and Devillers 1993 bioconcentration factor model is the best estimate of BCFs. This model was developed and tested using emperical data, and is supported by Devillers *et al.*, 1996 (*Chemosphere* 33:6 1047-1065).

However, to ensure a conservative estimate of risk, the USEPA *Final Water Quality Guidance for the Great Lakes System* (1995) guidance, which assumes BCF=K_{ow}, will be used.

B7 USEPA methodology provided in *Soil Screening Guidance: Technical Background Document*, 1996, will be used to convert log K_{ow} to log K_{oc}. The chosen equation best fits site related compounds, semivolatile nonionizing organic compounds.

B8 Agree in part. Soil samples derived from 0-6 feet will be assessed in the assessment of burrowing animals.

B9 Additional safety factors will be employed, as necessary, for endangered species. The text will be modified.

The scaling procedure described in the text uses EPA's *Draft Report: A Cross-Species Scaling Factor for Carcinogenic Risk Assessment Based on Equivalence of* $mg/kg^{3/4}/day$ as its basis.

Agree. If sediments fail to meet sediment benchmark values, additional studies may be required. However, this is not covered under the current scope of work.