Volume III - Ecological Risk Assessment

TNT Area B Remedial Investigation
Former Plum Brook Ordnance Works
Sandusky, Ohio

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List of Acronyms

AOC	area(s) of concern
ARP	assessment receptor profile
BAF	bioaccumulation factor
BCF	bioconcentration factor
BERA	baseline ecological risk assessment
BTAG	Biological Technical Assistance Group
COPEC	chemical(s) of potential ecological concern
DNAP	Ohio Division of Natural Areas and Preserves
DNT	dinitrotoluene
ESBSNP	Erie Sand Barrens State Nature Preserve
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
LOAEL	low observable adverse effects level
MDC	maximum detectable concentration
mg/kg	milligrams per kilogram
msl	mean sea level
NASA	National Aeronautic Space Administration
NOAEL	no observable adverse effects level
NWI	National Wetland Inventory
ODNR	Ohio Department of Natural Resources
OME	Ontario Ministry of the Environment and Energy
PBOW	Plum Brook Ordnance Works
RBSEV	risk-based screening ecotoxicity value
RTE	rare, threatened, or endangered
RTV	reference toxicity value
SQL	sample quantitation limit

List of Acronyms (Continued)

TF	transfer factor
TNT	trinitrotoluene
UCL	upper confidence limit
USFWS	U.S. Fish and Wildlife Service
UTL	upper tolerance limit

1.0 Introduction

A baseline ecological risk assessment (ERA[BERA]) has been performed to provide an estimate of current and future ecological risk associated with potential hazardous substance releases within trinitrotoluene (TNT) Area B at the former Plum Brook Ordnance Works (PBOW) in Sandusky, Ohio. The results of the BERA contribute to the overall characterization of the sites and serve as part of the baseline used to develop, evaluate, and select appropriate remedial alternatives, if necessary. The ERA has been 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 primary objective of the BERA 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 site, determining the particular hazardous substances being released from the site, identifying pathways for receptor exposure, and estimating the magnitude and likelihood of potential risk to identified receptors. The BERA addresses the potential for adverse effects to the vegetation, wildlife, aquatic life (aquatic macro-invertebrates), endangered and threatened species, and wetlands or other sensitive habitats associated with the site.

Concentrations of chemicals have been measured in relevant environmental media including soil, surface water, and sediment. No groundwater data have been collected for the TNT Area B site. Using available concentration data, IT Corporation (IT) has performed a BERA, including a problem formulation (Chapter 2.0); an exposure characterization (Chapter 3.0); an ecological effects characterization (Chapter 4.0); a risk characterization (Chapter 5.0); a risk summary and identification of preliminary remedial action objectives (Chapter 6.0); and conclusions and recommendations (Chapter 7.0). References are presented in Chapter 8.0.

IT has evaluated 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 has been conducted in two steps:

(1) a screening assessment step and (2) a predictive assessment step. Ecological endpoints to be addressed in both steps have been identified. The results and conclusions of the screening assessment determine whether a predictive assessment is required. The criteria by which a predictive assessment is needed have been 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:

- The 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.
- The potential for adverse ecological effects to ecological entities at the site is minimal or nonexistent due to the lack of **potential ecological receptors**.
- The potential for adverse ecological effects to ecological entities at the site is minimal or nonexistent due to the lack of **potential exposure pathways**.
- The 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 fourth null hypothesis is tested with the results of COPEC selection (Sections 2.2 and 2.3).

If a predictive assessment is triggered, terrestrial and aquatic ecological conceptual site models are developed, as appropriate, and additional problem formulation tasks are performed, as described in Sections 2.4 to 2.6.

2.1 Ecological Site Description

This ecological site description section includes a general discussion of site background and areas of concern (AOC), surface water resources, wetlands, vegetative communities, a species inventory, and a discussion on threatened and endangered species.

2.1.1 General Site Background

Plum Brook Ordnance Works (PBOW), approximately 6,453 acres in size, is located within the Eastern Lake Plains physiographic region of the Eastern Huron/Erie Lake Plain Ecoregion (Lafferty 1979; Omernik, 1986). This region is generally characterized as containing flat plains as the predominant land-surface form and as having a dominant natural vegetation of elm and ash in undisturbed areas. Approximately two-thirds of Erie County was once covered by a glacial

lake that produced features such as beach ridges and wave-cut cliffs. Much of the region is poorly drained due to the flat topography and low stream gradients. Many of the wetlands adjacent to Lake Erie in this region have been preserved by various federal, state, and private organizations (Peterjohn and Rice, 1991), thereby providing important wetland habitat for wildlife.

Across PBOW, the land slopes gently to the north-northeast towards Lake Erie. Elevations range from 675 feet above mean sea level (msl) at the southwest edge of the site to 625 feet msl in the northern portion of the property at Bogart Road, resulting in an average slope of approximately 0.3 percent. The Lake Plains region itself is over 69 percent cropland, 2.7 percent pasture land, and 10.5 percent forest (Ohio Department of Natural Resources [ODNR], 1985). However, since the Trojan Power Company (the previous PBOW owner) acquired the site in the early 1940s and removed the land from agricultural production, undeveloped portions of the station have become second generation forest and open fields. This has resulted in PBOW becoming an island of forest and open fields within a sea of agricultural land in north central Ohio.

Site soils associated with the TNT B Area include the Arkport-Galen soil association (Soil Conservation Services, 1971). Arkport soils are characterized as deep, nearly level to moderately sloping, well-drained to moderately-well-drained, and having a subsoil of loamy fine sand and fine sand. Galen soils are mostly level and moderately well-drained, with find sand or sandy loam surface layers, and a subsurface of fine sand which is underlain by silt or clay. The thickness of soils at PBOW ranges from approximately 5 feet or less for most of the site to approximately 20 feet for the extreme northern portion of the site (Erie Planning Commission Report on Land Capability, 1967 in Dames & Moore [1997]). The Arkport-Galen association is comprised of about 40 percent Arkport soils, 30 percent Galen soils, and 30 percent minor soils. Permeability of Arkport and Galen soils is estimated to be 12 and 6 to 12 inches per hour, respectively. Bedrock at PBOW includes Ohio (Huron) shale, Prout limestone, Plum Brook shale, and Delaware limestone (Erie Planning Commission Report on Land Capability, 1967 in Dames & Moore [1997]). The TNT B Area is underlain primarily by Plum Brook shale, and to a lesser extent by Delaware limestone.

AOC at the site include intermittent headwaters to Ransom Brook, standing water along North Magazine Road, and area wetlands. These AOC are discussed in more detail in Sections 2.1.2 through 2.1.3 in the following subsections. Based on a site reconnaissance performed by IT on

October 27th, 1998, a photographic record of the site was prepared and is presented in Figures 2-1 through 2-4. The site reconnaissance was performed by two IT ecologists. The site is undulating with low-lying areas that may contain standing water in the spring. There is a mix of open fields with low (3 to 4 inch) and medium (2 to 3 feet) vegetation, plus small woodlots. Prior to arrival at the site, IT personnel obtained relevant information on the site, including topographic maps; township, county, or other appropriate maps; and determined the location of potential ecological units such as streams, creeks, ponds, grasslands, forest, and wetlands on or near the sites. Additionally, the 1994 biological inventory of PBOW (National Aeronautics and Space Administration [NASA], 1995), which identifies and shows the locations of threatened and endangered species at PBOW, was reviewed. IT personnel completed a checklist similar to EPA's checklist for ecological assessment/sampling (EPA, 1997); and information from this checklist was used to complete Chapter 2.0 herein. The location of known or potential contaminant sources affecting the sites and the probable gradient of the pathway by which contaminants may be released from the site to the surrounding environment were identified. IT personnel used the reconnaissance to evaluate the sites for more subtle clues of potential effects from contaminant release.

2.1.2 Surface Water

The headwaters of Ransom Brook are the primary surface waters at the site, and were observed during the site visit to have limited flow (less than 5 gallons per minute) or were dry in some areas (Figure 2-2, Photo No. 4). The headwaters are classified as intermittent on the U.S. Geological Survey topographical map, and during the site visit were approximately 2 to 3 feet wide and several inches deep or dry in some areas. The substrate of the headwater flowages was typically bedrock or silty muck. The banks along the headwater flowages were generally low. North of the site, Ransom Brook flows into Pipe Creek. Standing water was observed north of the site along North Magazine Road, with duck weed floating on the surface (Figure 2-3, Photos Nos. 5 and 6). This standing water was 3 to 4 feet wide and 15 to 20 feet long. Based on the low-lying contours of the depression next to the road, it appeared the standing water could reach a depth of 2 to 3 feet during rainfall events. The source of the standing water was unknown. A storm drain on the south side of North Magazine Road was dry. The substrate of the standing water was muck and silt. The banks of the depressional area were moderately steep (about 3 to 4 feet from the road to the bottom of the depression). There was no unusual color or areas of coloration in either the headwaters of Ransom Brook or the standing water along North

Magazine Road. Given the nature of the surface waters at the site, they are not likely to support forage fish due to their shallow depth and intermittent nature.

Numerous small drainage ditches cross the site, but most were dry during the site visit.

2.1.3 Wetlands

According to the National Wetland Inventory (NWI) Maps for the area (U.S. Fish and Wildlife Services [USFWS], 1977), there are no designated wetlands at the TNT Area B site. It should be noted that the accuracy of NWI maps are limited, especially in relatively flat landscapes (such as Plum Brook Ordnance Works) because minor depressions often contain isolated wetlands not easily identified through air photo interpretation (the process used by the USFWS in preparing NWI maps). As discussed in the following section, small seasonal wetlands do exist at the site.

2.1.4 Vegetative Communities

Vegetative communities at the site were classified during the site reconnaissance trip using the 16 possible community types presented in Table 2-1. The 4 largest community types observed at the site were moderate old field (OFM), palustrine shrub/shrub wetland (PSS), early successional shrub thicket (ESU), and moderate successional shrub thicket (MSU), as shown in Figures 2-1 through 2-4. Wetland communities identified included PSS (listed above), plus small wet meadows and marshes (classified as palustrine emergent wetlands [PEM]). Wooded areas are generally restricted to just outside the boundary of the site, with oak trees dominant.

The general habitat figure (Figure 2-5) presents the type and extent of biological communities present within the site. The most common plant species found in the major vegetative communities are as follows:

Moderate old field (OFM)
Allegheny blackberry
Dewberry
Queen Anne's Lace

Shrub Thicket (ESU, MSU)
Gray dogwood
Dewberry
Hawthorn
White snakeroot

Wet Meadow (PEM)
soft rush
green bulrush
fragrant flat-topped goldenrod

Marshes (PEM)
Broad-leaved cattail

Scrub/Shrub Wetland (PSS)
Silky dogwood
Wetland sedges
Cottonwood

Each of these habitat types can be expected to support different wildlife species assemblages; however, given the close proximity of the habitats to each other, many of the species (discussed below) would be expected to spend some amount of time within each community type for foraging, resting, and loafing activities, depending on the season.

During the site reconnaissance, the study areas were examined for vegetative stress, including looking for plants displaying stunted growth, poor foliage growth, tissue discoloration, and a loss of leaf coverage. Possible vegetative stress was observed at one location, as evidenced by an area devoid of vegetation (bare area; Figure 2-4 [Photos Nos. 7 and 8]). This area contained a shattered rock substrate, which may have contributed to the lack of vegetation. Potential adverse impacts of chemical stressors on plant growth are discussed further in Section 5.1. Based on site reconnaissance information, it does not appear that significant ecological threats exist at the sites, as there is no definitive absence of biota or animal life in areas expected to support these ecological components.

2.1.5 Species Inventory

Based on information from ODNR (1995) and collected during the site reconnaissance, species lists were prepared for plants, mammals, birds, reptiles, amphibians, and fish (Tables 2-2 through 2-7). Of the plant 365 species documented at the 6,453 acre station by the ODNR (Appendix A), 41 of the common and frequently observed species at the site are listed in Table 2-2.

Of the 43 species of mammals that may be found in the region based on species range maps, white-tailed deer were observed onsite during site reconnaissance (Table 2-3). Numerous deer tracks were also observed during the site reconnaissance, and evidence of deer bedding was seen

(Figure 2-2, Photo No. 3). The ground cover and understory of the forested areas outside the site boundary showed significant overbrowsing by area wildlife, most likely by deer.

Of the 129 species of birds that may be found in the region based on species range maps, 105 species (81 percent) have been recorded at the station by the ODNR during their multiyear studies (Table 2-4). Three bird species were documented at TNT Area B during the site reconnaissance performed October 27, 1998 by IT. Of the species recorded by the ODNR, 49 were neotropical migrants and would not be expected to nest at the station. ODNR (1995) notes that of the top 50 bird species recorded at the station, only 6 were ground nesters and 3 others occasionally nest on the ground, suggesting that ground nesters are having problems at the station. Burning practices used by NASA at the station, as well as the large deer population that feeds on much of the ground cover, limits the ground cover available for nesting birds and results in increased predation for these species (ODNR, 1995). It should be noted that NASA has recently altered its burning practices, burning approximately one-third of the station lands every year, rather than one-half of the lands as was performed in the past (Peecook, 1998). The 15 most abundant bird species recorded at the station by the ODNR included American robin; red-winged blackbird; European starling; song sparrow; common grackle; field sparrow; American goldfinch; indigo bunting; blue jay; common yellowthroat; brown-headed cowbird; house wren; gray catbird; northern cardinal; and cedar waxwing.

Of the 14 species of reptile that may be found in the region based on species range maps, ten species (71 percent) have been observed at the station, including turtles and snakes (ODNR, 1995; Table 2-5). During the site reconnaissance no reptiles were observed.

Of the 10 species of amphibians that may be found in the region based on species range maps, nine species (90 percent) have been observed at the station (ODNR, 1995; Table 2-6), including salamanders, toads, and frogs. During the site reconnaissance a gray tree frog was observed.

Fourteen species of fish have been observed at the station, including suckers, sunfish, minnows, sticklebacks, and bullheads (Table 2-7). None are expected at TNT Area B given the limited surface water habitat.

In addition to the above species, additional species observed during site reconnaissance included praying mantis, wooly bear caterpillar, banana spider, and ants (active ant mounds).

2.1.6 Threatened and Endangered Species Information

According to an Ohio Division of Natural Areas and Preserves (DNAP) review of their natural heritage maps and files (Woischke, 1998), there are records of legal status threatened or endangered species within a two-mile radius of the site. These species include the following:

- Sedge wren (*Cistothorus platensis*) - endangered
- Dwarf bulrush (*Lipocarpha micrantha*) - threatened
- Twisted yellow-eye-grass (*Xyris torta*) - threatened
- Field sedge (*Carex conoidea*) - threatened
- Least St. John's-wort (*Hypericum gymnanthum*) - endangered
- Flat-leaved rush (*Juncus platyphyllus*) - endangered
- Bushy aster (*Aster dumosus*) - threatened.

In addition, based on information contained in ODNR (1995), there are several species of threatened or endangered plants, potentially threatened plants, and threatened or endangered birds that have been recorded at PBOW, as follows:

- Grove sandwort (*Arenaria lateriflora*) - threatened
- Thin-leaved sedge (*C. cephaloidea*) - endangered
- Ashy sunflower (*Helianthus mollis*) - threatened
- Prairie false indigo (*Baptisia lactea*) - potentially threatened
- Broad-winged sedge (*C. alata*) - potentially threatened
- Round-fruited hedge-hyssop (*Gratiola virginiana*) - potentially threatened
- Tall St. John's wort (*H. majus*) - potentially threatened
- Virginia meadow beauty (*Rhexia virginica*) - potentially threatened
- Tall nut rush (*Scleria triglomerata*) - potentially threatened
- Lance-leaved violet (*Viola lanceolata*) - potentially threatened
- Winter wren (*Trogoldytes troglodytes*) - endangered
- Cattle egret (*Bublucus ibis*) - endangered
- Black-crowned night heron (*Nycticorax nycticorax*) - threatened
- Trumpeter swan (*Cygnus buccinator*) - endangered
- Upland sandpiper (plover) (*Bartramia longicauda*) - threatened
- Indiana bat (*Mytolis sodalis*) - endangered.

Furthermore, wild white lettuce - a species considered extinct in Ohio but common in prairie states - was recently found on site, although several miles from TNT Area B (Peecook, 1998). Based on the results of the site reconnaissance, no threatened or endangered species of plants, birds, or mammals were found. The site reconnaissance also included detailed searches performed by a highly qualified botanist subcontractor during the October 27, 1998 site visit.

After a careful evaluation, IT's subcontractor botanist determined that it is unlikely that any rare, threatened, or endangered species of plants inhabit the site (Appendix B).

None of the threatened or endangered bird species, with the possible exception of the sedge wren (also called the short-billed marsh wren) and the black-crowned night heron, would typically be expected to be found at the site. The winter wren, cattle egret, trumpeter swan, and upland sandpiper are all considered rare visitors or migrants at the station (ODNR, 1995), and have not been documented nesting within a one-mile of the site (Woischke, 1998).

Six sedge wren nesting sites were documented in the ODNR DNAP's data base for an area near the Pentolite Road Red Water Ponds area in old grassy field habitat during 1994 (Woischke, 1998). In the June 1994 sighting, eight sedge wrens were observed near the Pentolite Road Red Water Ponds and two were observed near the West Area Red Water Ponds, with 24 territorial birds and several fledglings identified at the Plum Brook Ordnance Works (ODNR, 1995). Sedge wrens, an Ohio endangered species, have a preferred breeding habitat that consists of wet meadows, grassy marshes, and old grassy fields (Peterson, 1947) and moist grasses and sedge marshes or wet meadows with scattered low bushes (Bent, 1948). These habitat types are limited at TNT Area B, although there are a few wet meadows (Figure 2-5). Sedge wrens are very sporadic nesters, and birds may not arrive at a breeding colony until July or August and nesting continues into September (Trautman and Trautman, 1968). Draining of marshes has eliminated considerable breeding habitat in Ohio and many recent colonies have been located in wet meadows and hayfields and have disappeared after these fields have been mowed (ODNR DNAP information).

The black-crowned night heron, an Ohio threatened species, is a regular visitor at ponds, streams, and ditches within the station, however, it does not nest at the station (ODNR, 1995; Woischke, 1998). The species is typically found near water and wetlands, and since the early 1980s there has been a nesting colony of approximately 100 pairs located on an island in Sandusky Bay approximately 10 miles north northwest of the study area (Peterjohn and Rice, 1991).

The Indiana bat has not been documented at the site, and is generally not expected in the area based on the fact that its preferred habitat (e.g., caves along streams or trees with exfoliated bark) is not present at TNT Area B. Trees with exfoliated bark, such as shagbark or shellbark hickory, are rare or not present at the site, thereby providing little bat roosting habitat (Appendix A).

With the exception of the Erie Sand Barrens State Nature Preserve (ESBSNP), there are no existing or proposed state nature preserves or scenic rivers near the site, and ODNR is unaware of any unique ecological sites, geological features, breeding or nonbreeding animal concentrations, champion trees, or state parks, forests, or wildlife areas within a 2-mile radius of the site (Woischke, 1998). The ESBSNP is located west southwest of TNT Area B. The 32-acre preserve is a remnant sand beach of Lake Warren, the fifth ancestral Lake Erie, that supports many species of threatened and endangered plant species such as field sedge, Least St. John's wort, dwarf bullrush, twisted yellow-eyed-grass, flat-leaved rush, bushy aster, and Virginia meadow beauty. Many of the preserve's rare plant species thrive in open wind-swept conditions such as those found on the sand barrens. The DNAP actively manages the preserve to ensure that the open wind-swept areas remain and do not become overgrown with woody vegetation.

2.2 Selection of Chemicals of Potential Ecological Concern

IT has identified 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. These chemicals are also present at sufficient frequency, concentration, and location to pose a potential risk to ecological receptors. Examples of screening criteria that have been used include the following: analytical detection limit; frequency of detection less than 5 percent; comparability with background (Table 2-8); role as an ecologically essential nutrient at site concentrations; and comparison with risk-based screening ecotoxicity values. This selection process is described in more detail in the following subsections.

2.2.1 Data Organization

The data for each chemical have been sorted by medium. For ecological impacts, soil from 0 to 6 feet have been considered. Chemicals which are not detected at least once in a medium have not been included in the risk assessment. Available background data have been determined for each medium. Sources of background information include data from previous investigations.

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

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

- J - Value is estimated, probably below the contract-required quantitation limit.
- R - Quality control indicates that the data are unusable (chemical may or may not be present).
- B - Concentration of chemical in sample is not sufficiently higher than concentration in the blank (using 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 in Section 2.2.2.

2.2.2 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 have been estimated for each chemical in each medium of interest (Tables 2-9 through 2-12) . 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 (i.e., there are no chemical stressors at the site).

Data sets have been tested for normality and lognormality based on the Shapiro-Wilk test (EPA, 1992a). Statistical analysis has been performed on all detected chemicals. 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. If a data set passes both the normal and lognormal distribution tests, the distribution with the best fit is selected.

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

$$UCL = \bar{x} + t_{1-\alpha, n-1} \times (s/\sqrt{n}) \qquad \text{Eq. 2.1}$$

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):

$$UCL = e^{\left(\bar{y} + (0.5 \cdot s_y^2) + \left[H_{0.95} \cdot \frac{s_y}{(n-1)^{0.5}}\right]\right)} \qquad \text{Eq. 2.2}$$

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):

$$u = p(n + 1) + Z_{1-\alpha}\sqrt{np(1 - p)} \qquad \text{Eq. 2.3}$$

where:

u = upper confidence limit
p = percentile corresponding to the arithmetic mean (constituent specific)
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 maximum detected concentration (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.

2.2.3 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 have not been 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. For the current assessment, nitroaromatics have not been dropped as COPECs as this group of constituents is site related. Chemicals detected infrequently at high concentrations may identify the existence of "hot spots" and have been retained in the evaluation, unless other information exists to suggest that their presence is unlikely to be related to site activities.

2.2.4 Natural Site Constituents (Background and Essential Nutrients)

Chemical concentrations have been 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, besides PAHs, 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. For the current BERA, this more rigorous statistical analysis was performed for copper and thallium (see Human Health Risk Assessment Appendix C).

The UTL was calculated as follows:

$$UTL = \overline{X} + k(a) \qquad \text{Eq. 2.4}$$

where:

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

For background data that were determined to have a nonparametric distribution, based on use of the Shapiro-Wilk Test (EPA, 1992b), and for data sets with greater than 15 percent nondetects, the 95th percentile (rather than the UTL) was used, as discussed for soil in IT (1998). If the 95th percentile was a nondetect value, the maximum detected value was used. Results are presented in Table 2-8. Background surface water and sediment results were not available.

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. However, for the current BERA, this additional analysis was not performed.

Essential nutrients such as calcium, chloride, iodine, magnesium, phosphorus, potassium and sodium were eliminated as COPEC only if their maximum detected concentration did not exceed the background level, and provided that their presence in a particular medium was judged to be unlikely to cause adverse effects on wildlife. However, as there are no screening criteria that are readily available for nutrients, they have been retained as COPECs unless shown to be below the background UTL or 95th percentile.

2.2.5 Comparison to Risk-Based Screening Ecotoxicity Values

A comparison has been made between MDCs of chemicals in sampled media, and risk-based screening ecotoxicity values (RBSEV) for ecological endpoints following recommendations in EPA Region 5 BTAG Bulletin No. 1 (EPA, 1996a) and the ecological risk assessment work plan (IT, 1999). The following RBSEV have been used for the ecological screening assessment:

- ***Soil.*** Soil screening values from EPA Region 3 *BTAG Screening Levels* (EPA, 1995a), *Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Terrestrial Plants* (Will and Suter, 1995a); *Toxicological Benchmarks for Contaminants of Potential Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Process* (Will and Suter, 1995b) were used. It should be noted that effects on heterotrophic processes may not be relevant to ecological receptors of concern at the site.

- ***Groundwater.*** Groundwater was not collected from TNT Area B, thus RBSEVs for groundwater are not appropriate.

- ***Surface Water.*** EPA ecotox threshold (ET) screening values for freshwater (EPA, 1996b) and Ohio EPA water quality criteria were used.

- ***Sediment.*** EPA ET values for freshwater sediment and sediment criteria from the Ontario Ministry of the Environment and Energy (OME, 1993) were used, as recommended in EPA Region 5 guidance (EPA, 1996a).

- ***All Media.*** *Preliminary Remediation Goals for Ecological Endpoints* (Efroymson, et al., 1997), *Ecological Data Quality Levels, RCRA Appendix IX Hazardous Constituents* (EPA Region 5, 1998a) were used.

The results of the screening of the selected COPECs with RBSEV for ecological endpoints of concern are presented in Tables 2-9 through 2-12. COPECs are only selected for further consideration in the BERA if the MDC exceeded the available RBSEV. If no RBSEV was available, the constituent was carried forward for consideration in the BERA, unless it was within background or less than 5 percent occurrence. Note: Appendix C presents a compilation of the RBSEVs for each medium, and shows the lowest (i.e., most conservative) value that is used in the screening assessment.

2.2.6 Summary of COPEC Selection

Tables 2-9 through 2-12 have been prepared for each medium, with the following information:

- Chemical name

- Frequency of detection
- Arithmetic mean of site concentrations
- Range of detected concentrations
- Range of detection limits
- Statistical distribution
- UCL of the arithmetic mean
- Background screening criterion, if available
- Risk-based screening criterion, if available
- Exclusion rationale for selection as COPEC
- COPEC selection conclusion: YES or NO
- Source-term exposure concentration that may be used in the BERA.

Footnotes in the tables provide the rationale for selecting or rejecting a chemical as a COPEC. An evaluation of all of the constituents that were eliminated for further consideration was performed to determine whether any should be reinstated as COPECs due to other considerations. Examples of these exceptions include: potential break-down products; chemicals known to have been used onsite historically; chemicals with detection limits greater than the RBSEV; and chemicals with high bioconcentration and/or bioaccumulation factors. Based on this evaluation, no additional COPECs are recommended. Twenty-eight COPECs have been selected for surface soil, thirty-two have been selected for total soil, twelve have been chosen for surface water, and fourteen have been chosen for sediment. As discussed at the beginning of Chapter 2.0, given this selection of potential chemical stressors, and on the finding that viable habitat, potential receptors, and potential exposure pathways exist at the site, a predictive assessment is triggered. Chemicals not eliminated using the screening procedures previously presented are considered final COPECs and have been quantitatively evaluated in the predictive BERA. The relevant and important physical, chemical, and toxicological properties of the identified COPEC risk drivers have been reviewed from the scientific literature and are presented as COPEC profiles in Appendix D.

2.3 Selection of Assessment Receptors

IT has selected assessment receptors for evaluation during the predictive BERA. In order to focus the exposure characterization portion of the BERA on species or components that are the most likely to be affected, given the toxicological and mobility characteristics of the COPECs, and on those COPECs that, if affected, are most likely to produce greater effects in the on-site ecosystem, IT has focused the selection process on species, groups of species, or functional groups, rather than higher organization levels such as communities or ecosystems. Site biota are 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 at the sites was determined during a literature review and during the site reconnaissance (Section 2.1.7 and 2.1.8) prior to identification of target receptor species.

Primary criteria for selecting appropriate assessment receptors included, but were not 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 the COPECs given their mode of toxicity, propensity to bioaccumulate, etc.
- 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. Based on the availability of species-specific data, a RTE surrogate species may be selected (Section 2.3.1).

Additional criteria for selection of assessment receptors was used to identify species that offer the most favorable combination of characteristics for determining the implications of on-site contaminants. These criteria included: (1) limited home range; (2) role in local nonhuman food chains; (3) potential high abundance and wide distribution at the site; (4) sufficient toxicological information available in the literature for comparative and interpretive purposes; (5) sensitivity to COPECs; (6) relatively high likelihood of occurrence on-site following remediation (if required); (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.

2.3.1 Terrestrial Receptors

Seven representative receptor species that are expected or possible in the area of TNT Area B (Section 2.1) were selected as indicator species for the potential effects of COPECs. These indicator species represent two classes of vertebrate wildlife (mammals and birds) and a range of both body size and food habits, including herbivory, omnivory, and carnivory. Note: potential impacts to terrestrial plants are considered in Section 5.1. The seven species selected include the deer mouse (*Peromyscus maniculatus*) (small, omnivorous mammal), short-tailed shrew (*Blarina brevicauda*) (small, insectivorous mammal), Eastern cottontail rabbit (*Sylvilagus floridanus*) (medium-sized herbivorous mammal), marsh wren (*Cistothorus palustris*) (small insectivorous bird), white-tailed deer (*Odocoileus virginianus*) (large herbivorous mammal), raccoon (*Procyon lotor*) (medium-sized omnivorous mammal), and red-tailed hawk (*Buteo jamaicensis*) (large, carnivorous bird). The marsh wren was selected as a surrogate for the sedge wren, an Ohio endangered species that has been documented in the general area and a species that may be expected on site given the availability of some preferred nesting habitat.

The deer mouse, shrew, Eastern cottontail, and wren represent the prey base for the larger predators of the area (represented by the red-tailed hawk). A terrestrial food web is presented in Figure 2-6. Many of these species have limited home ranges, particularly the deer mouse, cottontail, shrew, and marsh wren, which make them particularly vulnerable to exposure to site contaminants. All of the selected terrestrial receptor species have a potential high abundance and wide distribution at the site and sufficient toxicological information (with the exception of some bird species) is available in the literature for comparative and interpretive purposes. In addition, all of the selected species are likely to occur after site remediation (if risk management decisions require it), and all are important to the stability of the local ecological food chain and biotic community. Finally, all the selected species have readily-available exposure data, as summarized in the *Wildlife Exposure Factors Handbook* (EPA, 1993).

Larger mammal species were generally not selected as sensitive receptors due to their large home ranges, however, the far-ranging red-tailed hawk was retained due to its unique role as a top predator in the food chain and the white-tailed deer was retained due to its high abundance at the site. Smaller birds were generally not included because most are migratory. The potential risk to species with larger home ranges and migratory avian species will be included within the predicted risks to the selected terrestrial indicator receptors. Foraging factors were conservatively set to 100 percent for the mouse, shrew, rabbit, wren, and raccoon, due to their relatively

small home ranges (Section 3.1). However, for the deer and hawk, the foraging factor was set at 0.05 and 0.03 (or 5 and 3 percent), respectively, based on these two species' relatively large home ranges (518 and 842 hectares, or 1,280 and 2,081 acres, respectively), compared with the size of the site (approximately 60 acres, Figure 2-5).

Results of the assessment receptor selection process are presented in detailed biological and ecological descriptions called assessment receptor profiles (ARP). Additionally, the biologically relevant criteria used to select the seven terrestrial assessment receptors are discussed and summarized in the ARP (Appendix E).

2.3.2 Aquatic Receptors

The only aquatic habitats at the site include the headwaters of Ransom Brook and standing water along North Magazine Road, and small isolated palustrine shrub/scrub and small wet meadows and marshes (palustrine emergent wetlands) (Sections 2.1.2 and 2.1.3; Figure 2-3). Exposure to aquatic organisms within the water bodies and/or wetlands is assumed to occur via direct exposure to contaminants in the water column and via ingestion of benthic invertebrates as well as littoral and pelagic prey exposed to contaminants in surface water and sediment. Note: potential effects to macroinvertebrates and phytoplankton (algae) have been assessed using available surface water and sediment quality criteria for the protection of aquatic life (Section 5.2). Potential uptake through the food chain is evaluated for two representative receptors, including the raccoon (also considered as a terrestrial receptor) and the mallard (*Anas platyrhynchos*) (medium-sized aquatic omnivore).

Aquatic organisms represent some of the prey base for aquatic receptors (represented by the mallard and raccoon). A food web is presented in Figure 2-7. The selected receptor species have relatively small home ranges, which makes them particularly vulnerable to exposure to site contaminants. Foraging factors were set to 20 percent for both of the aquatic receptors evaluated in this BERA because of the limited amount of surface water and sediment at the site (less than 1 acre, Section 2.1.2). Both of the selected aquatic receptor species have been documented near the site (Section 2.1), have a potential high abundance and wide distribution at the site, and sufficient toxicological information (with the exception of the mallard bird species) is available in the literature for comparative and interpretive purposes. In addition, both of the selected species are likely to occur after site remediation (if risk management decisions require it), and both are important to the stability of the local ecological food chain and biotic community.

Finally, the selected species have readily available exposure data, as summarized in the *Wildlife Exposure Factors Handbook* (EPA, 1993).

Results of the assessment receptor selection process are presented in detailed biological and ecological descriptions known as ARPs. The biological relevant criteria used to select the aquatic assessment receptors are discussed and summarized in the ARPs (Appendix E).

2.4 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 BERA. Key aspects of ecological protection are presented as policy goals. These are general goals established by legislation or agency policy that are based on societal concern for the protection of certain environmental resources. For example, environmental protection is mandated by a variety of legislation and government agency policies (e.g., CERCLA, National Environmental Policy Act). Other legislation includes the Endangered Species Act 16 U.S.C. 1531-1544 (1993, as amended) and the Migratory Bird Treaty Act 16 U.S.C. 703-711 (1993, as amended). To determine whether these protection goals are met at the site, assessment and measurement endpoints have been formulated to define the specific ecological values to be protected and to define the degree to which each may be protected.

Unlike the human health risk assessment process, which focuses on individual receptors, the BERA focuses on populations or groups of interbreeding nonhuman, nondomesticated receptors. In the BERA process, the risks to individuals are 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 should 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 are 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. Information gained during the site reconnaissance was used to assist in the selection of assessment and measurement endpoints. These endpoints, formal expressions of the environmental values to be protected (Suter, 1993), have been used to focus the goals of the BERA (Table 2-13).

Measurement endpoints for this BERA are 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 have been concurrently selected by identifying those that are known to be adversely affected by chemicals at the site based on published literature. COPECs for those receptors and endpoints have been identified by drawing on the scientific literature to obtain information regarding potential toxic effects of site chemicals to site species. This process ensures 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.

2.4.1 Assessment Endpoints

The assessment endpoints for the TNT Area B are stated as "the protection of long-term survival and reproductive capabilities for terrestrial invertebrates, herbivorous mammals, omnivorous mammals, insectivorous mammals and birds, carnivorous birds, benthic invertebrates, omnivorous aquatic mammals, and omnivorous aquatic birds." The corresponding null hypothesis (H_o) for each of the assessment endpoints is stated as: "the presence of site contaminants within soil, surface water, sediment, vegetation, and prey will have no effect on the survival or reproductive capabilities of terrestrial invertebrates, herbivorous mammals, omnivorous mammals, insectivorous mammals and birds, carnivorous birds, benthic invertebrates, omnivorous aquatic mammals, and omnivorous aquatic birds."

Assessment receptor species were selected based on the likelihood of finding the species at the TNT Area B. Historical information, the site reconnaissance (performed October 27, 1998), and the availability of toxicological data were used to select terrestrial and aquatic receptor species. These receptors species are depicted in food web models (Figures 2-6 and 2-7). Food web

models are simplified versions of the possible movement of contaminants through the food chain present or potentially present at the site. Due to lack of data for all possible species, key species have been selected to represent broad classes, or guilds.

The food web conceptual site models were developed to illustrate how the selected terrestrial and aquatic species are ecologically linked within food webs. One species was used to represent each of the major trophic levels and habitats at the site. The decision was made not to complicate the food web models with detailed species selection at the base of the food web (*i.e.* specific terrestrial/benthic invertebrates or aquatic vertebrates). Thus, generic terrestrial invertebrates, benthic invertebrates, and aquatic invertebrates were used to represent the bottom of the food chain. For terrestrial invertebrates and plants, partitioning coefficients and simple empirical uptake models were employed to estimate COPEC concentrations within tissues (Chapter 3.0). These tissue concentrations were then used as input values for exposure to higher trophic level receptors through the dietary ingestion route. Brief life-history descriptions for the selected area receptor species are provided in Appendix E).

All trophic levels may be exposed to COPECs, either by direct exposure to contaminated abiotic media or through ingestion of lower trophic level food items. Primary producers (plants) absorb COPECs (as well as nutrients) from soil and/or water. Through abiotic processes COPECs can adsorb to the sediment and detritus particles. When these particles settle and become part of the benthic substrate they may also become a source of COPECs to benthic communities. Various species of aquatic biota fulfill the role of aquatic herbivorous (feeding on aquatic plants and suspended detritus) and predatory invertebrates (feeding on benthic invertebrate species). The combination of COPEC bioconcentration from water, ingestion of contaminated prey, and generally restricted ranges for aquatic organisms provides ideal conditions for significant bioconcentration of COPECs. For this reason the mallard was included in the aquatic food web as a top trophic-level omnivore capable of bioaccumulating COPECs. In terrestrial species bioconcentration occurs in plants and invertebrates, and higher food chain receptors bioaccumulate COPECs through the ingestion of food items.

2.4.2 Measurement Endpoints

Measurement endpoints are frequently numerical expressions of observations (*e.g.* toxicity test results or community diversity indices) that can be compared statistically to detect adverse responses to a site contaminant. Examples of typical measurement endpoints include mortality,

growth or reproduction in toxicity tests; individual abundance; species diversity; and the presence or absence of indicator data in field surveys of existing impacts (EPA, 1994).

For assessments, measurable responses to stressors may include lowest observed adverse effect levels (LOAEL), no observed adverse effect levels (NOAEL), LC_{50}s (lethal concentration to 50 percent of the test population), LD_{50}s (lethal dose to 50 percent of the test population), or EC_{20}s (effective concentration for 20 percent of the test population), collectively termed "toxicity endpoint values." In addition, critical effect values for surface water, sediment, and soil were selected as measurement endpoints (Table 2-13). The most appropriate measurement endpoint(s) were chosen based on exposure pathways as well as ecotoxicity of the contaminant.

2.5 Ecological Site Conceptual Model

IT has prepared pictorial representations of potential exposure. These food web pictorials (Figures 2-6 and 2-7) and accompanying text presented in Section 3.1, clarify the representations that are the ecological site conceptual models (ESCM). The ESCMs trace the contaminant pathways through both abiotic components and biotic food web components of the environment. The ESCMs present all potentially complete exposure pathways. The ESCMs have been 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.

3.0 Exposure Characterization

IT has developed an estimate of the nature, extent, and magnitude of potential exposure of assessment receptors to COPECs 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 COPECs during the selection process (Section 2.3). The exposure assessment has been conducted by linking 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 has performed an exposure analysis, which combines the spatial and temporal distribution of the ecological receptors with those of the COPEC to evaluate exposure. The exposure analysis focuses on the chemical amounts that are assumed to be bioavailable, and the means by which the ecological receptors are exposed (e.g., exposure pathways). The focus of the analysis is dependent on the assessment receptors being evaluated as well as the assessment and measurement endpoints.

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 F).

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 such as grain size and organic carbon content.

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 have been addressed within this BERA. If any of these components are not complete, then contaminants in those media do not constitute an environmental risk at that specific site. As discussed in Section 2.3, however, these four components are complete for all sampled site media. The major fate and transport properties associated with typical site contaminants are presented in subsequent sections. These properties directly affect a contaminant's behavior in each of the exposure pathway components.

For terrestrial faunal receptors, calculation of exposure rates relies upon determination of an organism's exposure to COPECs found in surface water, surface soil, and on transfer factors used for food-chain exposure. Exposure rates for terrestrial wildlife receptors are 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 has not been estimated. In addition, dermal and inhalation 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 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 have also been gathered on incidental ingestion of soil, and are incorporated for the receptor species. This information is summarized in Table 3-1.

Algorithms have been evaluated for calculating exposure for terrestrial vertebrates that account for exposure via ingestion of contaminated water, incidental ingestion of contaminated soil,

ingestion of plants grown in contaminated soil, and prey items. Singular algorithms have been developed for soil to plant uptake and for animal bioaccumulation (transfer factors).

The basic equation for estimating dose through the dietary pathway is:

$$D_p = \sum_{k=1}^{m} (C_k \cdot F_k \cdot I_k) / W$$

where:

D_p = the potential average daily dose (mg/kg-day),
C_k = the average COPEC concentration in the kth food type (mg/kg dry weight)
F_k = the fraction of the kth food type that is contaminated
I_k = the ingestion rate of the kth food type (kg dry weight/day)
W = the body weight of the receptor (kg wet weight).

For aquatic faunal receptors, the calculation of exposure rates depends on the determination of the contaminant concentration in water and sediment, and on food-chain multipliers, bioconcentration factors (BCF), and bioaccumulation factors (BAF). If appropriate, an evaluation can be made of the time each organism spends associated with surface water or sediment pore water in order to modify exposure rates, however, this refined approach was not used in the current BERA.

Literature values for animal-specific sediment ingestion have been used if available. However, such values generally are not available in the literature. Where sediment ingestion rates could not be found, the animal-specific incidental soil ingestion rate is 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 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's (1993) algorithm to calculate pore water concentrations for nonionic organic chemicals was used, as follows:

$$\text{Pore water concentration (milligrams per liter)} = (SC)/[(F_{oc})\,(K_{oc})] \qquad \text{Eq. 3.1}$$

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).

For sediment, K_{ow} values (Table 3-2) have been converted to K_{oc} values (EPA, 1996c) as follows:

$$\log K_{oc} = 0.00028 + (0.983 \times \log K_{ow}) \qquad \text{Eq. 3.2}$$

where:

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

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

F_{oc} values for sediment have been set at 0.067, based on sediment samples collected from the West Area Road Red Water Pond (IT, 1999).

To estimate inorganic partitioning from sediment to pore water, K_d values (Table 3-3; with units of L/kg) were used instead of F_{oc} and K_{oc}. It should be noted that estimates of inorganic COPEC concentrations in sediment pore water were set at the calculated value using the appropriate K_d or were set at the metal's published solubility in water, whichever was lower. This approach was necessary as some of the common nutrients' estimated pore-water concentrations exceeded typical solubility values because inorganic partitioning using K_d values is inaccurate for nutrients with elevated concentrations. Nutrient solubilities were assumed to be best represented by the common environmental compounds MgO, CaO, Al_2O_3, $FeCO_3$, and ZnO.

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

$$BCF = K_{ow} \qquad \text{Eq. 3.3}$$

where:

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

When possible, K_{ow} values for appropriate COPEC have been obtained from the literature or from databases, and are presented in Table 3-2.

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.

Per EPA (1995b) guidance, aquatic BAFs have been 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 has been 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 BERA, soil-to-plant and food-to-muscle BAFs (or transfer factors [TF]) have been estimated for organic constituents using the log K_{ow} relationships developed by Travis and Arms (1988), as presented below, with calculated transfer factors presented in Appendix G.

$$\log Tf_{soil\text{-}to\text{-}plant} = 1.588 - 0.578(\log K_{ow})$$
$$\log Tf_{food\text{-}to\text{-}muscle} = -7.735 + 1.033(\log K_{ow}).$$

Soil-to-insect BAFs are based on log K_{ow} relationships developed by Connell and Markwell (1990). For organic COPECs in soil invertebrates, the transfer factor was derived from the following equation developed by Connell and Markwell (1990) for bioaccumulation in earthworms:

$$BAF = \frac{y_L \cdot K_{ow}^{b-a}}{x \cdot f_{oc}}$$

where:

BAF = the bioaccumulation factor (unitless)
y_L = the fractional lipid content of the organism
K_{ow} = the octanol/water partition coefficient
(b-a) = a nonlinearity constant
x = a proportionality constant
f_{oc} = the fractional organic carbon content in the soil.

Although derived from earthworm data, the values for the nonlinearity constant (0.05) and proportionality constant (0.66) were applied to modeling uptake in soil invertebrates. Because of differences in integument, it is expected that the uptake by earthworms will generally be greater than that of invertebrates such as insects. Therefore, these factors are expected to yield conservative estimates of invertebrate uptake. The lipid content in insects was estimated at 3.1 percent fresh weight (Taylor, 1975), which is 7.9 percent of dry weight, using a value of 61 percent water content in beetles (EPA, 1993), calculated as follows:

$$\frac{0.031}{(1 - 0.61)} = 0.079 \text{ or } 7.9 \text{ percent}$$

The fraction of organic carbon in the soil was estimated to be 0.01, or 1 percent, as site-specific soil f_{oc} data were not available. This default value was based on information related to fate and transport data for organic constituents, as presented in *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (EPA, 1998b). Except where literature-derived values are available, the soil-to-invertebrate transfer factors for inorganics were assumed to be 1. Table 3-3 presents the soil-to-plant and soil-to-invertebrate transfer factors estimated for the inorganic COPECs. The COPECs in this table are limited to those that were not dropped during the screening assessment (Section 2.3). Transfer factors for organic COPECs are presented in the risk characterization spreadsheets presented in Appendix G.

Tissue concentrations in vertebrate prey species were estimated from the daily intake of the COPECs through the use of transfer factors for beef. The regression equation developed by Travis and Arms (1988) was used to derive food-to-beef transfer factors for the organic COPECs based on the log K_{ow} value of the chemical of concern. Transfer factors for the inorganic COPECs were taken from International Atomic Energy Agency (1994), National Council on Radiation Porotection and Measurements (1989), Ma (1982), and Baes, et al. (1984), as shown in Table 3-3. A weighted total of the concentrations of COPEC intakes (including ingested soil and surface water) was then used in the calculation of tissue concentrations in prey species and the dietary exposure rate in all selected receptor assessment species, as follows:

$$\frac{\text{Total intake of soil, water, plants, and/or invertebrates (in mg COPEC/day)} \times \text{Food-to-Tissue TF}}{0.32 \times \text{Total food and soil intake (in kg mass/day)}}$$

A conversion factor of 0.32 was used to convert wet weight tissue concentrations to dry weight values, given that the water content of mammals and passerine birds is reported to be 68 percent (Table 4-1 in EPA, 1993).

Media-Specific Exposure Pathways. Exposure to four categories of environmental media are addressed in this BERA, 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 have been considered, as this would be the point of exposure. For burrowing animals such as the shrew, soil samples obtained from a depth of 0 to 6 feet have been considered.

For deep-rooted plant exposure, soil samples taken from 0 to 6 feet have been considered because most feeder roots are located within this depth. Thus, the white-tailed deer is assumed to ingest leaves of trees translocating COPECs from subsoils (Figure 2-6).

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 soil contaminants have been used (Table 3-3).

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 and pesticides) decreases with increasing concentrations of total organic carbon in the sediment; however, these compounds can still bioaccumulate up the food chain (Landrum and Robbins, 1990).

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

contaminants constitutes a potential indirect exposure pathway for faunal receptors. Piscivorous receptor's exposure to fish has not been quantified due to the lack of fish habitat at the site. Chemical bioavailability of some metals and other chemicals is controlled by water hardness, pH, and total suspended solids.

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

Groundwater itself is not an exposure point. However, contaminant transport along the shallow groundwater pathway would be considered an exposure route to aquatic life, wetlands, and some wildlife where the groundwater discharges to surface water. This pathway is of importance to aquatic and wetland receptors if groundwater is found to be discharging to surface water. A groundwater assessment is not included in this BERA because no groundwater samples were collected. In addition, groundwater discharge to surface water at TNT Area B is not suspected, based on existing data.

3.2 Exposure Characterization Summary

The estimated chemical intakes for each exposed receptor group under each exposure pathway and scenario are presented in the risk characterization spreadsheets in Appendix G. These intake estimates are combined with the COPEC toxicity values, discussed in the following section, to derive estimates and characterize potential ecological risk. The uncertainties associated with the estimation of chemical intake are discussed in Section 5.4). The basis for each uncertainty has been identified, with the degree of uncertainty estimated qualitatively (low, medium, or high) or quantitatively, and the impact of the uncertainty estimated qualitatively (overestimate or underestimate, as appropriate).

4.0 Ecological Effects Characterization

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

4.1 Selection of Literature Benchmark Values

IT has consulted appropriate sources for literature benchmark values, such as (1) *Toxicological Benchmarks for Wildlife* (Sample, et al., 1996); and (2) LD_{50} values from data bases such as the Registry of Toxic Effects Concentrations (RTEC) (extrapolated to chronic NOAEL or LOAEL values using recommended Tri-Service [Wentsel, 1996] uncertainty factors). The level of effort has been limited to documents that summarize the available ecotoxicological information and does not consist of a review of the primary toxicological literature (i.e., IT has not reviewed details of toxicity test conditions to determine validity of the tests performed).

4.2 Development of Reference Toxicity Values

IT has developed or determined reference toxicity values (RTV) for the site. These RTVs focus on the growth, survival, and reproduction of species and/or populations. Empirical data is available for the specific receptor-endpoint combinations in some instances. However, for some COPECs, data on surrogate species and/or on endpoints other than the NOAEL and LOAEL had 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 may 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 had to be used. Safety factors are 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 has been gathered for those analytes identified as COPECs. Because the measurement endpoint ranges from the NOAEL to the LOAEL, preference has been 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., benzo[a]pyrene for other PAHs) have been gathered for use in the BERA.

Using the relevant toxicity information, RTVs have been calculated for each of the COPECs. RTVs represent NOAELs and LOAELs with safety factors incorporated for toxicity information derived from studies other than no-effects or lowest-effects studies. The hierarchy employed by this BERA for selection of RTVs is:

- NOAELs,
- LOAELs (corrected to NOAELs by applying a safety factor of 10),
- LD_{50} (corrected to NOAELs by applying a safety factor of 100).

For studies on species other than the receptors selected for this risk assessment, additional uncertainty factors have been used. RTVs have been calculated using safety factors specified in Wentsel, et al. (1996) and shown in Figure 4-1 and Table 4-1. Interclass toxicity extrapolations were not performed as physiological differences between classes are too great to be addressed with the use of simplistic safety factors. Separate factors are used to account for extrapolation to the no effects or lowest-effects endpoints, for study duration (Table 4-1), and for extrapolation across taxonomic groups (e.g., species, genus, family, order), as shown in Table 4-2 for the receptors used in this BERA. Although additional safety factors may be employed for endangered species, no endangered species were selected as representative receptors and these additional safety factors were not required. Because NOAELs for the selected wildlife receptor species are based on NOAELs from test species, the latter are 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) and shown below. A body weight scaling factor of 0.25 was used for mammals, whereas a body weight scaling factor of 0 was used for birds.

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

where:

$NOAEL_W$ = the No Observed Adverse Effect Level for the wildlife indicator species (mg/kg-day)
$NOAEL_T$ = the No Observed Adverse Effect Level for the test species (mg/kg-day)

BW_T = the body weight of the test species (kg)
BW_W = the body weight of the wildlife indicator species (kg)
s = a body weight scaling factor (s = 1/4 for mammals and s = 0 for birds).

Test species body weights (BW_T) used for COPEC RTVs are contained in the risk characterization spreadsheets presented in Appendix G.

These factors were used together to derive a final adjusted RTV, as shown in the risk characterization spreadsheets (Appendix G). Reference toxicity threshold value uncertainties are discussed in Section 5.4.

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 are calculated, or as in the case of plant receptors, exposure is simply calculated as the soil concentration (Section 5.1).

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. Qualitative and semiquantitative approaches have been taken to estimate the likelihood of adverse effects occurring as a result of exposure of the selected site receptors to COPECs. Potential adverse affects to terrestrial plants have been qualitatively assessed by comparing plant toxicity benchmarks with COPEC concentrations. Potential adverse impacts to aquatic biota have been qualitatively assessed by comparing surface water and sediment quality criteria for the protection of aquatic life with surface water and sediment COPEC concentrations.

For the semiquantitative predictive assessment, RTVs and exposure rates have been calculated and are 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 adverse toxicological effects will occur among site receptors.

5.1 Terrestrial Plant Impact Assessment

To assess the potential impact of COPEC concentrations in surface soil on terrestrial plant species, the source-term concentrations from TNT Area B were compared with available benchmark concentrations developed for the protection of terrestrial plants. As shown in Table 5-1, only two benchmarks were exceeded by the COPEC source-term concentrations (for lead and selenium). It should be noted, however, that few benchmarks were readily available for the COPECs retained for the predictive BERA. Although one bare soil area was noted during the site reconnaissance (Section 2.1.4), this finding may not be attributed to chemical stressors in surface soils for the following two reasons: (1) the maximum concentrations of lead and selenium occurred in Sample No. 10730 near Bldg. 456 (245 milligrams per kilogram [mg/kg] for lead) and in Sample No. 10500 near Bldg. 417 (3.9 mg/kg for selenium), and neither of these buildings are near the location of the bare soil area discussed in Section 2.1.4 (location east of grid point B-8 near Bldg. 472 shown in Photos Nos. 7 and 8 (Figure 2-4); and (2) the area of bare soil is closest to surface soil Sample No. 10890, which is recorded as having lead and selenium concentrations of 35 and 2.4 mg/kg, respectively. As the lead and selenium benchmarks are 50 and 1.0 mg/kg, respectively (Table 5-1), the elevated level of selenium is a

concern for vegetative stress, but it is possible that the bare area may be related to some other factor, such as the substrate type.

5.2 Aquatic Biota Impact Assessment

To assess the potential impact of COPEC concentrations in surface water and sediment on aquatic biota, the source-term concentrations from the TNT Area B were compared with available benchmark concentrations developed for the protection of aquatic life. As shown in Table 5-2, surface water COPEC concentrations for aluminum, calcium, copper, iron, lead, manganese, and selenium exceeded some or all of the available benchmarks for the protection of aquatic life. As up to four different benchmarks are used for the surface water assessment, a weight-of-evidence approach may be taken, with more significant impacts being predicted as the number of surface water benchmarks that are exceeded increases. Thus, for TNT Area B, elevated concentrations of calcium are not a concern because only one surface water benchmark was exceeded. The six other metals had a source-term concentration that exceeded two of the four benchmarks. It is important to note that the limited number of surface water samples collected at the TNT Area B resulted in the maximum measured concentration for each of the inorganics being used as the source-term concentration (Table 2-11). It is possible that additional sampling would result in different conclusions for the impact assessment. It is also important to note that surface water is limited at TNT Area B, and is not a major habitat type.

As shown in Table 5-3, sediment COPEC concentrations for 2-methlynaphthalene, nickel, and zinc measured in the TNT Area B exceeded the majority of the available benchmarks developed for the protection of aquatic biota. It should be noted that EPA Region 5 (1996) BTAG recommends that National Oceanic and Atmospheric Administration (NOAA) effects range-low and effects range-medium benchmarks (Table 5-3) only be used when freshwater benchmarks are unavailable, as NOAA benchmarks are primarily for marine environments. Discounting of the NOAA benchmarks, however, would not change the above findings. It is important to note that sediment is limited at TNT Area B, and is not a major habitat type.

5.3 Predictive Risk Estimation for Terrestrial and Aquatic Wildlife

IT has estimated the risk associated with TNT Area B. The risk estimation has been performed through a series of quantitative HQ calculations that compare receptor-specific exposure values with RTVs. The HQs are 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 exposure; and HQs greater than 100 present the highest potential for expected effects (Wentsel, et al., 1996). It should be noted that OEPA considers HQs greater than 1.0 to be potentially significant.

The simple HQ ratios have been summed to provide conservative hazard index estimates for all chemicals and exposure pathways for a given receptor. The following criterion may be used to determine if HQ summation is appropriate and scientifically defensible: for a given receptor, only HQs for those chemicals that have a similar mode of toxicological action should 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. The summation of HQs into a hazard index was performed, in part, to determine whether or not individual COPEC HQs should be segregated based on dissimilar modes of toxicological action. If a risk driver resulted in an HQ greater than approximately 10 to 100, segregation of other COPECs by mode of toxicological action would not be necessary.

Conservative hazard indices (summed HQs) for terrestrial receptors at the TNT Area B are 4,970 for the deer mouse, 1,2341,239 for the cottontail rabbit, 18,006 for the shrew, 40,308 for the marsh wren, 4,372 for the raccoon, 67 for the white-tailed deer, and 3 for the red-tailed hawk (Table 5-4). COPECs from soil were risk drivers for the deer mouse, cottontail, shrew, marsh wren, raccoon, and hawk, while COPECs from water were the risk drivers for the deer receptor. Risk drivers, listed in decreasing order of concern, included 2,4,6-TNT, 4-amino-2,6-dinitrotoluene (DNT), 2-amino-4,6-DNT, and Aroclor-1260 from soil, and aluminum and iron from surface water. Important routes of exposure included invertebrate and plant intake and direct surface water intake (Table 5-4).

Conservative hazard indices (summed HQs) for aquatic receptors at the TNT Area B are 56 for the mallard and 6 for the raccoon (Table 5-5). It should be noted that both terrestrial and aquatic hazards have been estimated for the raccoon as it exhibits a combined terrestrial and aquatic lifestyle, and hazards may be summed for this receptor. COPECs from surface water and sediment were risk drivers for the raccoon, while COPECs from sediment were the risk drivers for the mallard receptor. Risk drivers (i.e., HQs greater than 1.0), listed in decreasing order of concern, included 2-amino-4,6-DNT and zinc from sediment, and aluminum from water.

Important routes of exposure included aquatic invertebrate intake and direct surface water intake (Table 5-5).

5.4 Uncertainty Analysis

The results of the BERA are 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, and field studies using site-specific receptor species). Detailed media, prey, and receptor field studies are costly; thus, the preliminary predictive analyses of risk has been conducted to limit the potential use of these resource-intensive techniques to only those COPECs that continue to show a relatively high potential for ecological risk. Since assessment criteria were developed based on conservative assumptions, the results of the screening and predictive assessments 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.

The uncertainty analysis is presented in Table 5-6 and lists some of the major assumptions made for the BERA; 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 BERA progresses to higher level assessment phases (EPA, 1992b). The uncertainty analysis identifies and, if possible, quantifies the uncertainty in the individual preliminary scoping assessment, problem formulation, exposure and effects assessment, and risk characterization phases of this BERA. Based on this uncertainty analysis, the most important biases, that may result in an overestimation of risk, include the following:

- Assuming that COPECs are 100 percent bioavailable
- Using laboratory-derived or empirically-estimated partitioning and transfer factors to predict COPEC concentrations in plants, invertebrates, and sediment pore water
- Using K_{ow} values without adjustment to estimate macroinvertebrate BCFs for organic COPECs.

It should be noted that the maximum 2,4,6-TNT concentration in surface soil (6,900 mg/kg) was used as the source-term concentration (due to the undefined data distribution), while the arithmetic mean concentration was 261 mg/kg. This estimate of central tendency is 26 times lower than the maximum concentration and suggests that using the maximum concentration is overly conservative. It should also be noted that the elevated hazards predicted for the marsh wren for 4-amino-2,6-DNT and 2-amino-4,6-DNT were based on an adjusted LD_{50} value for 4-amino-2-nitrotoluene for an unspecified bird species (Table 4-1) and are of high uncertainty. It should also be noted that the maximum aluminum and iron concentrations measured in surface water were used as the source-term concentrations due to the limited number of samples (Table 2-11) and an additional sampling effort could potentially reduce the hazard estimate.

It should be noted that the maximum concentrations of many of the COPECs measured in surface water and sediment were used as source-term concentrations due to the limited number of samples, and an additional sampling effort could potentially reduce the hazard estimate. It should also be mentioned that there are significant uncertainties associated with estimating concentrations in macroinvertebrates (Section 5.4).

5.5 Risk Description

As part of the risk description, IT has completed the following: (1) summarized the ecological risk associated with the site; and (2) interpreted the ecological significance, which describes the magnitude of the identified risks and the accompanying uncertainty. The effect of additional

data or analyses on uncertainty has also been discussed. A weight-of-evidence approach has been used to interpret the ecological significance of the findings.

Soil COPEC impacts to terrestrial plants are estimated to be generally insignificant, except for perhaps elevated levels of lead and selenium (Section 5.1). Surface water COPEC impacts to aquatic biota are of most concern for aluminum, copper, iron, lead, manganese, and selenium (Section 5.2). Sediment COPEC impacts on aquatic biota are of potential concern for 2-methyl-naphthalene, nickel, and zinc (Section 5.2). Considerations of limited sample size suggest these findings should be confirmed based on additional sampling before any remedial actions are considered. In addition, limited aquatic habitat at the site reduces the concern for impact to aquatic biota. Terrestrial receptors (especially mice, rabbits, shrews, and wrens) are predicted to incur elevated hazards from exposure to 2,4,6-TNT, 4-amino-2,6-DNT, 2-amino-4,6-DNT, and Aroclor-1260 in soil and from exposure to aluminum and iron in surface water (Section 5.3). Based on uncertainties of toxicity, and on the fact that no RTE species have been confirmed at the site, remedial actions may not be warranted at this time for soil.

Aquatic receptors are predicted to have elevated hazards from exposure to 2-amino-4,6-DNT and zinc from sediment, and aluminum from water (Section 5.3). Based on uncertainties associated with toxicity, estimating concentrations in aquatic insects, and limited sample size, no remedial actions are warranted at this time for surface water or sediment.

6.0 Risk Summary and Identification of Preliminary Remedial Action Objectives

IT has summarized ecological risk associated with releases from the site (Section 5.5). This summary is supported by tasks performed during the previous sections. Additionally, IT has made recommendations for further risk investigation, if appropriate. IT has not developed site-specific remedial action objectives for the site at this time due to previously discussed uncertainties associated with the BERA (Section 5.4).

7.0 Conclusions and Recommendations

Only the data, results, and conclusions of the various preliminary scoping and predictive assessment phases are described in this section. No recommendations concerning types of remedial actions to be conducted are given other than to present the specific remedial action objectives, if any. Conclusion and recommendations are derived from the risk assessment and are based on the responses to the assessment hypotheses. The predictive assessment results are summarized and presented in Tables 5-1 through 5-5. The predictive assessment suggests potential adverse impacts to terrestrial wildlife, especially mice, rabbits, shrews, raccoons, and birds like the marsh wren, for modeled contact with the hazard drivers (2,4,6-TNT and two DNT-isomers) in surface soil.

The predictive assessment results may serve as the foci of discussions with risk managers and regulatory agencies concerning the potential need for additional assessment at PBOW TNT Area B to reduce the uncertainty in the estimate of ecological risk, as summarized in Table 5-6. It is very important to note that many conservative assumptions and modeling approaches were used in the predictive assessment, and actual hazards to wildlife may be orders of magnitude lower than predicted herein.

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