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**FINAL**

**2002-2003  
SITE MANAGEMENT PLAN**

**NAVAL WEAPONS STATION YORKTOWN  
YORKTOWN, VIRGINIA  
AND  
CHEATHAM ANNEX SITE  
WILLIAMSBURG, VIRGINIA**

**CONTRACT TASK ORDER 0195**

**MAY 2002**

*Prepared for:*

**DEPARTMENT OF THE NAVY  
ATLANTIC DIVISION  
NAVAL FACILITIES  
ENGINEERING COMMAND  
*Norfolk, Virginia***

*Under the:*

**LANTDIV CLEAN Program  
Contract N62470-95-D-6007**

*Prepared by:*

**CH2M Hill  
*Herndon, Virginia***

**BAKER ENVIRONMENTAL, INC.  
*Coraopolis, Pennsylvania***

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## LIST OF ACRONYMS AND ABBREVIATIONS{tc \11 "LIST OF ACRONYMS AND ABBREVIATIONS}

AET	Apparent Effect Threshold
AOC	Area of Concern
ASTM	American Society for Testing and Materials
ATSDR	Agency for Toxic Substance and Disease Registry
ATV	All-Terrain Vehicle
AUF	Area Use Factor
AVS	Acids Volatile Sulfides
BAF	Bioaccumulation Factors
Baker	Baker Environmental Incorporated
BCF	bio-concentration factor
BHC	Benzenehexachloride
bgs	below ground surface
BTAG	Biological Technical Assistance Group
CAX	Cheatham Annex Site
CDI	Chronic Daily Intake
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CLP	Contract Laboratory Program
CFR	Code of Federal Regulations
CNO	Chief of Naval Operations
COC	Chemical of Concern
COPC	Chemical of Potential Concern
cPAH	Carcinogenic Polyaromatic Hydrocarbon
CRAVE	Carcinogen Risk Assessment Verification Endeavor
CRDL	Contract Required Detection Limit
CRQL	Contract Required Quantitation Limit
CSF	Cancer Slope Factor
CT	Central Tendency
CTO	Contract Task Order
DAD	Dermally Absorbed Dose
DDD	Dichlorodiphenyldichloroethane
DDE	Dichlorodiphenyldichloroethylene
DDT	Dichlorodiphenyltrichloroethane
DoN	Department of the Navy
ECOC	Ecological Chemical of Concern
EqP	Equilibrium PArTioning
ERA	Ecological Risk Assessment
ER-L	Effects Range-Low
ER-M	Effects Range-Median
ESE	Environmental Science and Engineering
FISC	Fleet and Industrial Supply Center
foc	fraction of organic carbon
FS	Feasibility Study
FSAP	Field Sampling and Analysis

## LIST OF ACRONYMS AND ABBREVIATIONS

(Continued)

GPS	Global Positioning Study
HASP	Health and Safety Plan
HEAST	Health Effect Assessment Summary Tables
HHRA	Human Health Risk Assessment
HI	Hazard Index
HQ	Hazard Quotient
IAS	Initial Assessment Study
ID	Inside Diameter
IDW	Investigation Derived Waste
IEUBK	Integrated Exposure Uptake Biokinetic
ILCR	Incremental Lifetime Cancer Risk
IRIS	Integrated Risk Information System
IRP	Installation Restoration Program
$K_{oc}$	Organic Carbon Adsorption Coefficient
$K_{ow}$	Octanol/Water Partition Coefficient
$K_d$	Adsorption Coefficient
$K_p$	Partitioning Coefficient
LANTIDV	Atlantic Division, Naval Facilities Engineering Command
LD <sub>50</sub>	Median Lethal Dose
LOAEL	Lowest Observable Adverse Effect Levels
MACT	Maximum Acceptable Toxicant Concentrations
MCL	Maximum Contaminant Level
mg/kg	Milligram per Kilogram
MI	Mobility Index
MSL	Mean Sea Level
MS/MSD	Matrix Spike/Matrix Spike Duplicate
NAD	North American Datum
NAVD	North American Vertical Datum
NAWQC	National Ambient Water Quality Criteria
NCEA	National Center for Environmental Assessment
NCP	National Oil and Hazardous Substances Pollution Contingency Plan
NEESA	Naval Energy and Environmental Support Activity
NFESC	Naval Facilities Engineering Service Center
NFG	National Functional Guideline
NOAEL	No Observed Adverse Effect Levels
NWS	Naval Weapons Station
OSWER	Office of Solid Waste and Emergency Response
PAH	Polynuclear Aromatic Hydrocarbon
PCB	Polychlorinated Biphenyl
PMCL	Primary Maximum Contaminant Level
ppm	Parts per Million
ppt	Parts per Thousand
PVC	Polyvinyl Chloride

## LIST OF ACRONYMS AND ABBREVIATIONS

(Continued)

QA/QC	Quality Assurance/Quality Control
QAPP	Quality Assurance Project Plan
RA	Risk Assessment
RAGS	Risk Assessment Guidance for Superfund
RBC	Risk-Based Concentration
RfC	Reference Concentration
RfD	Reference Dose
RI	Remedial Investigation
RME	Reasonable Maximum Exposure
RPD	Relative Percent Difference
S	Water Solubility
SA	Surface Area
SARA	Superfund Amendments and Reauthorization Act
SCCRBS	Selection of Contaminants of Concern by Risk-Based Screening
SDWA	Safe Drinking Water Act
SEM	Simultaneously Extracted Metals
SERA	Screening-Level Ecological Risk Assessment
SI	Site Investigation
SOP	Standard Operating Procedure
SSP	Site Screening Process
SSSL	Surface Soil Screening Level
SSSV	Surface Soil Screening Value
SSV	Sediment Screening Value
SVOC	Semi-Volatile Organic Compound
SWSL	Surface Water Screening Level
SWSV	Surface Water Screening Value
TAL	Target Analyte List
TBM	Temporary Bench Mark
TCE	Trichloroethylene
TCL	Target Compound List
TCLP	Toxicity Characteristic Leaching Procedure
TCRA	Time-Critical Removal Action
TOC	Total Organic Carbon
TPH	Total Petroleum Hydrocarbon
UCL	Upper Confidence Limit
USGS	United States Geological Survey
USEPA	United States Environmental Protection Agency
USN	United States Navy
VDEQ	Virginia Department of Environmental Quality
VOC	Volatile Organic Compound
VP	Vapor Pressure
VSWCB	Virginia State Water Control Board
WOE	Weight of Evidence
WPNSTA	Naval Weapons Station
WQP	Water Quality Parameters

## **1.0 INTRODUCTION**

This document presents the results of the Remedial Investigation (RI) which was conducted at Site 1 - Landfill near Incinerator, at Naval Weapons Station Yorktown, Yorktown, Virginia, Cheatham Annex Site (CAX), Williamsburg, Virginia, by Baker Environmental, Inc. (Baker). The Field Investigation (November 1999, and May 2000) was performed under contract to the Atlantic Division, Naval Facilities Engineering Command (LANTDIV), Contract Number N62470-95-D-6007, Contract Task Order (CTO) Number 0104. Additional field activities were conducted in August and November 2001.

In 1980, Congress enacted the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) establishing the Superfund Program to respond to releases and the threatened release of hazardous substances. CERCLA, as amended by the Superfund Amendments and Reauthorization Act (SARA) of 1986, required certain revisions to the National Oil and Hazardous Substances Pollution Contingency Plan (NCP) to implement the new authorities and responsibilities of the CERCLA amendments. The revisions to the NCP established assessment programs to investigate releases such as those established in Section 300.420, which specifies the site assessment process known as the pre-remedial process which designates sites for long-term remedial evaluation and response.

### **1.1 Remedial Investigation Objectives**

The objectives of this RI are threefold:

1. To delineate the lateral and vertical extent of the landfill (including a waste volume estimate), characterize wastes buried at the site, as well as the underlying native soil, and groundwater downgradient of the landfill.
2. To evaluate the nature and extent of the threat to public health and the environment caused by the release or threatened release of hazardous substances, pollutants, or contaminants.
3. To provide data required for establishing feasible alternatives for consideration during preparation of the Record of Decision (ROD).

The RI was conducted through the sampling of environmental media (i.e., soil, groundwater, surface water, and sediment) evaluating the resultant analytical and geologic data; and performing a qualitative assessment of the findings.

## **1.2 RI and RI Report Organization**

The remedial investigation includes data-generating field activities conducted in 1999, 2000, and 2001 which consisted of the following:

### 1999 Activities -

- Ten direct-push borings through the landfill (1-DPB01 through 1-DPB10)
- Five hand auger borings (99-EXP01 through 99-EXP05)
- Split-spoon sample verification at 5 of the "DPB" series borings above

### 2000 Activities -

- Installation of two groundwater monitoring wells (1-GW09 and 1-GW10)
- Groundwater sample collection (wells 1-GW05, 1-GW06, 1-GW07, 1-GW09, and 1-GW10)
- Six split-spoon borings (00-PB01 through 00-PB06)
- Geotechnical analysis of two soil samples (1-GEO01 and 1-GEO02)

### 2001 Activities -

- Ten surface soil samples (with two duplicate samples) (CX01-SS01 through CX01-SS10-00)
- Five soil boring samples, including one duplicate sample (CX01-SB01 through CX01-SB04)

- Five surface water samples, including one duplicate sample (CX01-SW01 through CX01-SW04)
- Twenty-two sediment samples, with two duplicate samples, (CX01-SD01-01 through CX01-SD11-01 and CX01-SD01-02 through CX01-SD11-02)

Data evaluation for this RI report will consider data from other investigations as well. This additional data will be used to supplement the 1999, 2000 and 2001 investigation data. This additional data includes surface and subsurface soils from 1992, as well as surface soils, subsurface soils, sediment, and surface water from 1998. The data from 1992, 1998, 1999, 2000 and 2001 have been integrated and evaluated to meet the objectives outlined in Section 1.1.

This RI report is organized into ten sections, including this section. The other sections include:

- Section 2.0 - Site History and Previous Investigations
- Section 3.0 - Environmental Setting
- Section 4.0 - Remedial Investigation Field Activities
- Section 5.0 - Nature and Extent of Contamination
- Section 6.0 - Contaminant Fate and Transport
- Section 7.0 - Human Health Risk Assessment
- Section 8.0 - Ecological Risk Assessment
- Section 9.0 - Conclusions and Recommendations
- Section 10.0- References

Tables and figures appear at the end of each section to which they pertain. Appendices appear at the end of the report and include support documentation, such as well construction records and sample chain-of-custody forms.

## **2.0 SITE HISTORY AND PREVIOUS INVESTIGATIONS**

This section describes the physical setting and history of CAX. In addition, a brief summary of previous environmental investigations conducted at Site 1 is presented.

### **2.1 History and Background Information**

The subsections that follow present summaries of the history and setting of CAX and specific information regarding Site 1 - Landfill near Incinerator.

#### **2.1.1 Cheatham Annex Location and History**

CAX is located on the south bank of the York River within York County, Virginia. CAX is situated northeast of Interstate 64, approximately 1 to 2 miles northeast of Williamsburg, Virginia. At inception in 1943, CAX occupied approximately 3,349 acres. Several portions of the original base have since been declared surplus and transferred to other government jurisdictions, including the National Park Service, the Commonwealth of Virginia, and York County. CAX is currently comprised of 1,578 acres. The base, or Activity is divided into two separate parcels, with the larger parcel situated along the banks of the York River. Almost all of the activities at CAX (administration, training, maintenance, support, and housing) take place in this portion of the Activity. The smaller parcel is located south of the Colonial National Historic Parkway (Colonial Parkway). This area contains the Activity's water supply (Jones Pond) and is used mainly as a watershed protection area. CAX and the surrounding properties are shown on Figure 2-1.

CAX was established in June 1943 as a satellite unit of the Naval Supply Depot to provide bulk storage facilities. Prior to 1943, CAX had been the location of the Penniman Shell Loading Plant, which was a large powder and shell loading facility operated by DuPont during World War I. Under contract to the United States Government, the Penniman Shell Loading Plant was built to support the war effort. The contract stipulated that part of the shell loading plant was to be completed by April 1918, with the entire plant to be fully operational by December 1918. When the Armistice was signed on November 11, 1918, the Penniman shell loading facility began to shut down, without becoming fully operational. By 1926, when DuPont disposed of the property, most of the buildings associated with the plant had been removed. From 1926 to 1943 the land was used for farming. Since 1943, CAX has been used for receiving, storing, packaging, and

shipping materials to federal facilities on the East Coast and to major distribution centers in Europe.

Previously operated as an annex to the Fleet and Industrial Supply Center (FISC), Norfolk, CAX provided logistic and supply support to Naval shore installations. CAX is the Navy Sea System Command's East Coast consolidated stock point for major shipboard mechanical, electronic, and some navigational equipment. In addition to receiving, storing, issuing, packing and shipping Navy stock material, particularly large shipboard equipment (e.g., submarine periscopes, ship propellers, bull gears, antennae, and sonar domes), CAX provides warehouse and distribution services for 39 Storage Authorization Programs and tenant organizations.

In July 1987, CAX was designated the Hampton Roads Navy Recreational Complex. Today the mission of CAX includes supplying Atlantic Fleet ships and providing recreational opportunities to military and civilian personnel; 55% of CAX is undeveloped and rich in natural resources. Outdoor recreational facilities and activities available include: 13 cabins, 19 recreational vehicle (RV) sites, camp sites, an 18-hole golf course, swimming pool, ball fields, fishing, boating, wildlife watching and hunting (Department of the Navy [DON], 1998). CAX currently operates under the command of Naval Weapons Station (WPNSTA) Yorktown. The transition of CAX control from FISC to WPNSTA occurred in October 1998.

### **2.1.2 Site 1 - Landfill Near Incinerator Location and History**

Site 1 is located along the York River behind the former location of the old incinerator (Figure 2-2). The incinerator has been dismantled. Although the exact date is unknown, it is believed that the incinerator was dismantled sometime between 1989 and 1992. The incinerator building is indicated on figures included in the Remedial Investigation Interim Report (Dames and Moore, 1989), and the Site Investigation (conducted in 1992) states that the incinerator has been removed. From 1942 to 1951 the landfill was used as a disposal area for burn residues (e.g., ash) and from 1951 to 1972 as a general landfill. A variety of wastes, including empty paint cans and paint thinner cans, cartons of ether and other unspecified drugs, railroad ties, tar paper, sawdust, rags, concrete, and lumber were burned and disposed in the landfill until 1981. After this time, the landfill was no longer used. A percentage breakdown of the waste types is unknown (Naval Energy and Environmental Support Activity [NEESA], 1984).

The Initial Assessment Study (IAS) conducted in 1984 estimated that 34,500 tons of solid waste were buried at the landfill. This estimate was based on limited information and is now known not to be accurate. Based on more current information, the landfill occupies an area of approximately one acre and the northern area of impacted soils occupies an area of approximately 0.3 acre. An estimated 14,750 total cubic yards of solid waste was buried at the landfill. This volume estimate is discussed further in Section 4.9. A new waste volume estimate is one of the objectives of this RI.

The surface of the landfill is relatively flat and is overgrown with vegetation during most of the year. The landfill was closed in 1981 by regrading, placing a 2-foot soil cover upon the debris and vegetating the soil cover. A fence encloses a portion of the landfill and vehicular access to this area by unauthorized personnel is restricted by a locked gate. There is no debris or other materials on the surface of the landfill within the fenced-in area. There is however, a large debris pile present north of the fenced-in area, as discussed below. The fence was installed as part of a government training activity unrelated to the landfill, and does not correspond with the landfill perimeter. Very rugged terrain and dense vegetation prevent access to portions of the landfill outside of the fence. Portions of the fence were taken down in 1998 to accommodate the geophysical survey conducted during the October 1998 Field Investigation.

The location of the landfill perimeter that is shown on Figure 2-2 is estimated and was determined based on interpretation of a landfill closure drawing (dated March 10, 1981) and review of aerial photographs presented in the Aerial Photographic Analysis (United States Environmental Protection Agency [USEPA], 1998a). The boundary was further delineated during the Field Investigation performed in 1998 (Baker, 1999) via geophysical survey and shallow confirmatory soil borings, and from the November 1999 and April 2000 direct push investigations conducted under this RI. Further delineation efforts were conducted in November 2001 through trenching activities to obtain more information on the nature and extent of landfill materials (Baker, 2002). The soils to the north of the actual landfill have been re-worked and contain varying amounts of debris such as charred materials, glass, metal and wood. It is likely that the debris was mixed in with the soil during placement of the soil cover. A large debris pile is present within this northern area which is not considered part of the actual landfill. The area contains cables, conex boxes, an empty storage tank, automobiles, airplane/boat parts, and other miscellaneous items. This area was previously designated as Area of Concern (AOC) 5 - Debris Area, but is currently being managed as part of Site 1. Landfill contents (including metal scrap, wood, drums, containers and

other miscellaneous debris) are exposed along portions of the western perimeter of the landfill along the edge of the marsh associated with the unnamed tributary to the York River.

There is a steep drop to the York River and adjacent creek at the edges of the open flat area. The areas immediately adjacent to the former landfill are wooded. The bank of the York River adjacent to the landfill is extremely steep (nearly vertical in areas), and is not vegetated. The York River is located approximately 25 feet below the landfill area at the bottom of the steep slope. During a visit to the site in March 1999 Baker noted that a small area along the northeastern perimeter was undergoing erosion and it appeared that landfill contents might be slowly washing into the York River. In this area, a thin layer of debris was exposed. A small rusty bucket, which contained an unidentified yellow substance, was present within the bank. Small clumps of ash/incinerator residue (and other debris, which apparently originated from the landfill) are sparsely present on the beach.

## **2.2 Previous Investigations and Actions**

The list that follows is a summary of investigations/studies related to Site 1 that have been conducted to date at CAX. It should be noted that sample names and locations are discussed further in Section 4.0 where appropriate.

- IAS of Naval Supply Center, Cheatham Annex and Yorktown Fuels Division dated February 1984 by NEESA.
- Confirmation Study, Step 1A (Verification Round One), June 1986 by Dames and Moore.
- Confirmation Study Step 1A (Round Two), dated June 1988 by Dames and Moore.
- Draft Remedial Investigation (RI) Interim Report dated March 1989 by Dames and Moore.
- Final RI Interim Report dated February 1991 by Environmental Science and Engineering (ESE).

- Final Site Investigation (SI) Investigation for Sites 1, 10, and 11 dated November 1994 by Baker and Weston.
- Site Screening Process Report, Sites 1, 10, and 11 dated September 1997 by Baker.
- Aerial Photographic Analysis. United States Navy (USN) Supply Center - Cheatham Annex, Williamsburg, Virginia. USEPA, Region III. May 1998.
- Shoreline Erosion Assessment Letter Report - Site 1 Landfill near Incinerator dated 1998 by Baker.
- Recommendations for Erosion Mitigation Measures Letter Report (Site 1). May 1999 by Baker.
- Action Memorandum for the Site 1 Time-Critical Removal Action (TCRA) dated August 1999 by Baker.
- Final TCRA Close-Out Report - June 2000 by Baker.
- Trenching Letter Report - Site 1, Site 4 and AOC 2. June 2002 by Baker.
- Draft Background Investigation Report - July 2002 by Baker.

In July 1983, LANTDIV collected a round of groundwater samples from each of the four monitoring wells that were installed as part of the site closure (1-GW01 through 1-GW04). The sampling, which was apparently not part of any formal investigation, included analysis for purgeable organics, engineering parameters, and select metals. Results were compared to groundwater quality standards and criteria. Total organic carbon, phenolics, iron, lead and zinc were elevated and pH was outside of the acceptable range. Iron, lead and zinc levels were not unexpected due to the galvanized steel well casing. Two additional rounds of groundwater sampling were recommended.

Due to the nature of the wastes disposed at the site (including paints, paint thinners, ether and unspecified drugs), the IAS recommended additional study for Site 1 to investigate potential contamination of groundwater and the York River.

The Confirmation Study Step 1A (Verification Round One, Dames and Moore, 1986) included installation of two new monitoring wells (1GW05 and 1GW06) and collection of groundwater samples from the four existing wells and two newly installed wells.

The Confirmation Study Step 1A (Round Two, Dames and Moore, 1988) included collection of an additional round of groundwater samples from the six monitoring wells in 1987.

The Final Remedial Investigation Interim Report (Dames and Moore, 1991) summarized the findings of the Confirmation Study. Potentially site-related contaminants detected in the Confirmation Study groundwater samples included oil and grease, and total phenols. The report recommended the site for further investigation to better define the nature and extent of contamination at the site. Recommended efforts included aerial photographic analysis, collection of an additional round of groundwater samples, and performance of a risk assessment.

The SI for Site 1 (Weston, 1994) included installation of two new monitoring wells (1GW07 and 1GW08), with collection of soil samples from both monitoring well borings. Groundwater samples were collected from each of the existing and newly installed monitoring wells, with the exception of well 1GW01, which was dry. A total of six sediment samples were collected from three sampling stations (two samples per station). Samples of ash/soil exhibited elevated levels of metals, polychlorinated biphenyls (PCBs), and total petroleum hydrocarbons (TPH), and detectable levels of semi-volatile organic compounds (SVOCs). The volatile organic compounds (VOCs) 4-methyl-2-pentanone and trichloroethylene (TCE), and TPH and metals were detected in groundwater, and the sediment samples contained low levels of TCE, SVOCs, TPH and metals. The Site Investigation concluded that a major release of contaminants to groundwater had not occurred and that as most of the debris was adequately covered, no immediate response or further investigation was required. The report did recommend re-sampling of monitoring wells for VOCs and dissolved (filtered) metals and a file search of past records to verify closure status of the landfill.

The Weston recommendations were not submitted to the Virginia Department of Environmental Quality (VDEQ) or the USEPA and additional investigations were conducted at Site 1. Weston did not sufficiently address the exposed drums and debris along the landfill perimeter, or the large debris pile, thus LANTDIV requested additional investigations.

In August 1997, Baker collected groundwater samples from seven of the eight monitoring wells at Site 1. These samples were collected as part of the Site Screening Process (SSP) investigation (Baker, 1997). Well 1GW03 could not be located at the time of the investigation. The samples were analyzed for Target Compound List (TCL) organics and total and dissolved Target Analyte List (TAL) metals. No organic compounds were detected. Concentrations of total (unfiltered) metals were significantly lower in the 1997 samples than in previously collected samples (e.g., Weston's 1994 investigation). This was due to the employment of low-flow sampling during the SSP investigation. Certain metals were detected at elevated levels. The SSP also included human health and ecological risk screening to determine whether contaminants detected in environmental media pose unacceptable risks to human receptors and/or the environment. The risk screening process was completed in accordance with the SSP Guidelines (Baker, 1994), and included previously collected soil and sediment samples.

The following is a summary of the conclusions/results that were presented in the SSP Report for Site 1:

- Based on the available analytical data, no unacceptable human health or ecological risks are posed by the site.
- Based on the available analytical data, no additional investigation or remedial action is warranted.
- The soil cover of the landfill should be maintained - trees that are growing through the cover should be removed.
- Monitoring wells that penetrate the landfill should be abandoned to eliminate a future potential pathway of contaminants from the fill material.

The SSP was designed to focus on groundwater as recommended by the SI. The SSP recommendations were based on risk screening. LANTDIV requested that additional investigation be conducted to address remediation of exposed debris, high levels of lead in waste materials, and erosion along the York River. Additionally, many of the existing wells were constructed with inappropriate materials (i.e., wells 1-GW01 through 1-GW04 used galvanized steel) or inadequately located (i.e., well 1-GW07 was located in landfill material and wells 1-GW06 and 1-GW08 were located hydraulically side-gradient).

Baker conducted an additional field investigation in 1998. This study included a geophysical survey to define the lateral extent of the landfill, and limited soil, sediment, and surface water sampling. Elevated levels of contaminants (primarily polynuclear aromatic hydrocarbons [PAHs] and heavy metals) were detected in soil and sediment. For Site 1, the Field Investigation Report (Baker, 1999) recommended the following:

- Remove surficial debris that has collected on the flat, inter-tidal beach area in the vicinity of the eroding bank.
- Develop and implement interim measures that can be quickly installed to mitigate erosion in the 60-foot stretch of shoreline where debris was exposed. (Baker developed recommendations for the interim erosion control measures for the small area of exposed debris at Site 1 in the letter report submitted May 6, 1999). The recommendations included use of sand-filled geo-textile tubes as a shoreline revetment. Construction should be implemented as a TCRA.
- Institute a periodic inspection program so that the condition of the slope can be monitored and documented.
- Consider abandoning monitoring wells 1GW01, 1GW02, 1GW03, 1GW04, and 1GW07 due to the installation of these monitoring wells through the landfill and/or monitoring well integrity.
- Develop and implement solutions for long-term management of the landfill.

Trees adjacent to the bank of the York River were removed in March 1999 to help prevent further erosion of the bank.

A TCRA for Site 1 was conducted in late 1999 and early 2000 to remove the debris that had collected on the beach area and to temporarily stabilize the toe of the bank in the erosion area. Toe stabilization was accomplished by installation of three sand-filled geotextile tubes. Upstream and downstream revetments were also installed along the York River to prevent erosion of the flanks. The TCRA was implemented to stabilize the site until the long-term solution for the management of the Site 1 landfill is implemented.

### **3.0 ENVIRONMENTAL SETTING**

This section presents a summary of information regarding the environmental setting of CAX, including geography, meteorology, surface water hydrology, geology, hydrogeology, ecology, land use, and demography. Specific hydrogeologic conditions encountered at the site are discussed in Section 4.0 (Remedial Investigation field Activities).

#### **3.1 Climate and Meteorology**

The moderating effects of the Atlantic Ocean influence the climate of the Virginia Peninsula. This results in mild winters and long, warm summers. High humidity frequently occurs along the coast and less frequently inland. Ground fog is frequent in the late summer, especially during the early morning hours. Freezing temperatures occur intermittently from October through March. Average monthly temperatures in the area range from approximately 38.8°F in January to 77.4°F in July.

Because of its location near the coastline, York County is subject to easterly storms throughout late summer and early fall, causing high tides and flooding. Intense hurricanes occasionally sweep the coast. Winter is characterized by storms that move along the eastern seaboard. The storms from the north are associated with high winds and precipitation occasionally in the form of snow, ice pellets, or rain; however, the snow is seldom prolonged or heavy. The average annual precipitation is 44.15 inches, with the summer months being the wettest and the winter months being the driest.

Spring is a period of contrasting weather, particularly during March. Spring and autumn are periods of frost. Summer is warm and humid with occasional showers and afternoon thunderstorms. Autumn is a season of comfortable temperatures (average temperature 60°F to 81°F) and generally pleasant weather.

Winds are highly variable in the area of CAX. Prevailing winds are usually from the south-southwest, but north-northeasterly winds are common in some months. Onshore winds predominate during the spring and summer.

### **3.2 Soil Associations**

Two soil types predominate at Site 1, namely Loamy Udorthents and Udorthents-Dumps complex (Figure 3-1). Loamy Udorthents consists of deep, well and moderately well drained loamy soil the have been disturbed by excavation and grading. The permeability of these soils is moderately rapid (2.5 to 5 inches per hour) to slow (0.05 to 0.2 inches), and are extremely to strongly acidic (a pH below 5.5 standard units). The Udorthents-Dumps complex consists of shallow to deep, excessive to moderately well drained soils in areas that have been disturbed during excavation and partially filled with garbage, trees, metal, flyash, or dredgings. The permeability of these soils is moderately rapid to slow, and are extremely to strongly acidic.

### **3.3 Regional Geology**

The geology of the Atlantic Coastal Plain physiographic province is characterized by unconsolidated sediments of Cretaceous, Tertiary, and Quaternary ages (see Table 3-1) that dip gently eastward and rest on pre-Cretaceous-aged bedrock at a depth of approximately 1,900 feet (VWCB, 1973). The bedrock consists primarily of crystalline igneous and metamorphic rock and scattered Triassic ("red bed") sedimentary rocks.

As discussed in the IAS for CAX (NEESA, 1984), the surficial unconsolidated sediments at CAX have been mapped as the Windsor Formation of the Pleistocene series. This formation is composed of a series of sands and silts deposited in marine and estuarine environments. Its thickness is estimated to vary from 0 to 40 feet at CAX. Underlying the Windsor Formation are the Miocene deposits of the Yorktown, St. Mary's, and Calvert Formations. The Miocene deposits range in thickness from approximately 200 feet in western James City County to slightly more than 300 feet in the Hampton area. The top portion of the Yorktown Formation, consisting of shells and shell fragments cemented with calcite, was encountered during the drilling of monitoring wells installed during Round One of the Confirmation Study and also during the SI. The Yorktown Formation grades downward into the St. Mary's Formation, which is composed of fine-grained, subround-to-round, quartz-grained sand, with a decrease in shell fragment content. The St. Mary's Formation has a darker color and is often called blue sand or blue clay in drilling logs. Underlying the St. Mary's Formation is the Calvert Formation. The base of the Calvert Formation is marked by a marl or coquina (NEESA, 1984).

The Eocene deposits, underlying the Miocene deposits, consist of the Chickahominy and Nanjemoy Formations. These formations consist of fine-to-medium-grained sand, with varying concentrations of glauconite. The thickness of the Eocene deposits varies considerably; in the vicinity of CAX, the thickness ranges from approximately 80 to 120 feet (VDMR, 1973, Plate 9).

Paleocene deposits, underlying the Eocene deposits, consist of the Aquia and Mattaponi Formations. In the central part of the York-James Peninsula, these formations consist of fine-grained quartz sand with 10 to 25% glauconite and numerous, relatively thin, silty clay stringers. These formations are approximately 100 feet thick in the central part of the York-James Peninsula (VWCB, 1973).

Cretaceous deposits of the Mattaponi (Lower) and Potomac Group Formations underlie the Paleocene deposits and constitute the lowermost unconsolidated sediments of the area. The Cretaceous deposits are characterized by discontinuous sand bodies interbedded with silts and clays. In the York-James Peninsula, these deposits are found in a fluvial-deltaic environment. The fluvial deposits are characteristically channel sand bodies that are coarse-grained at the base and become finer-grained upward. The deltaic deposits are medium-grained, moderately sorted sands. The Cretaceous deposits in the vicinity of CAX are approximately 1,450 feet thick (NEESA, 1984).

### **3.4 Hydrogeology**

The Atlantic Coastal Plain sediments are the most important source of potable water in the region. Recharge to the groundwater system is derived from precipitation. Approximately 50 percent of the precipitation is lost to evapotranspiration. The remaining 50 percent either results in surface runoff, or infiltrates and is introduced into the groundwater regime. Recharge of aquifers may occur at the surface near outcrop zones, or from downward migration from overlying strata.

The shallow aquifer is relatively thick at CAX. The groundwater study by the Virginia Water Control Board (VWCB, 1973) generalized the shallow aquifer system as the water table aquifer. The United States Geological Survey (USGS) conducted a groundwater study at the neighboring Naval Weapons Station Yorktown (USGS, 1997). The general water table aquifer was refined in this study, to include the following six units:

1. The Columbia aquifer
2. The Cornwallis Cave confining unit
3. The Cornwallis Cave aquifer
4. The Yorktown confining unit
5. The Yorktown-Eastover aquifer
6. The Eastover-Calvert confining unit

Hydrogeologic units are recognized only where they are saturated (for aquifers) or confining (for confining units). For example, although the strata that typically comprise a given aquifer (when saturated) are present, the hydrogeologic unit does not exist in areas where the unit is not saturated. Vertical migration of groundwater is typically impeded in areas where the confining units are continuous, relatively thick, and comprised primarily of low-permeability strata such as clay or silt. This aquifer is a source of domestic (older individual homes) water supplies in some parts of Charles City, New Kent, James City, and York Counties. According to the Health Department, the majority of the newer wells are below 200 feet in depth.

Historical data from monitoring wells installed throughout CAX can be used to infer occurrence of groundwater and general flow patterns. Groundwater generally occurs at depths less than 30 feet below ground surface (bgs) throughout upland areas of CAX and shallower in areas close to surface water bodies. The groundwater flow direction within the water table aquifer is generally toward surface water bodies. Therefore, the water table elevations roughly parallel surface topography. Seasonal variations in groundwater flow direction within the water table aquifer were not evident based upon winter 1986 and fall 1987 data. Findings from the SI confirm previous findings (Weston, 1994).

Based on information presented by Weston (Weston, 1994), the upper confined aquifer consists of the Calvert, Chickahominy, and Nanjemoy Formations. The bluish clayey St. Mary's Formation, which is approximately 100 feet thick, overlies the Calvert Formation and functions as an aquifer between the upper confined aquifer and the water table aquifer. The upper confined aquifer is generally 50 to 80 feet thick and consists of medium-grained sand, moderately-to-poorly sorted with glauconite, usually called green sand or black sand. The depth to the upper artesian aquifer is approximately 250 feet below MSL in the vicinity of CAX. The aquifer is a potential source of domestic water supply. Much of the recharge to the aquifer is probably derived from silts and clays of the St. Mary's Formation. Specific capacities of wells completed in this system range from 1 to 10 gallon/minute/foot.

Additionally, the principal artesian aquifer (the deepest of the three aquifers) consists of deposits of the Mattaponi and Potomac Group Formations of the Lower Cretaceous series and several discontinuous sand bodies interbedded with silt and clay. The top of the aquifer is approximately 450 feet below MSL in the vicinity of CAX. Recharge to the aquifer occurs through the outcrop in Henrico, Hanover, and western King William Counties. However, substantial recharge also occurs east of these areas from vertical leakage between the adjacent aquifers through the confining layers; it has been estimated at 30,500-gallons/day/square mile (gpd/mi<sup>2</sup>) of area. Transmissivities in the central and eastern parts of the aquifer (including CAX) vary from 15,000 to 50,000 gallons/day/foot. Flow direction is generally eastward toward the Chesapeake Bay. The most extensive development of the aquifer has occurred in the Richmond metropolitan area. Dissolved solids in the water increases with depth in an easterly direction and result in limited use of the aquifer east of Williamsburg, where total dissolved solids range from 1,500 to 9,000 parts per million (ppm) and chlorides may exceed 1,000 ppm. The deep aquifer is unusable as a potable water source at CAX because of its naturally poor quality (high hardness, elevated amounts of both total dissolved solids and fluorides).

### **3.5 Topography**

CAX is located in Williamsburg, Virginia, on the York-James Peninsula, which is an embayed portion of the Atlantic Coastal Plain physiographic province (VWCB, 1973). This elongated peninsula trends northwest-southeast and occupies an area of approximately 1,752 square miles. The peninsula is roughly bordered on the southwest by the James River, on the northeast by the York River, and on the southeast by the confluence of the James River and the Chesapeake Bay. At CAX, the peninsula is approximately 6 miles wide.

The topography is characterized by gently rolling terrain dissected by ravines and stream valleys trending predominantly northeastward toward the York River. Ground elevations at CAX vary from sea level along the eastern boundary, which borders the York River, to a maximum elevation of approximately 90 feet above MSL on a few scattered hills in the western portion of the site. Valleys consisting of 40- to 60-foot ravines with steep slopes (slopes exceeding 1:1) occur along the major creeks draining CAX (see Figure 3-2).

Site 1 is situated near the York River with steep banks along the York River and the unnamed tributary. The bank along the York River is approximately 15 to 20 high, and approximately 5-

feet high along the unnamed tributary. A narrow beach and broader tidal flat are present immediately north of the steep bank to the York River. The landfill surface slopes to the north from an elevation of approximately 23-feet above MSL to approximately 10-feet above MSL. A topographic high occurs southeast of the fenced area at an elevation of approximately 27-feet above MSL.

### **3.6 Surface Hydrology**

As discussed in the IAS for CAX (NEESA, 1984), the station lies entirely within the York River Basin. This basin, the central and eastern sections of Virginia, is located between the Rappahannock River Basin to the north and the James River Basin to the south. The headwaters rise in Orange County and flow approximately 120 miles in a southeasterly direction to the Chesapeake Bay. At CAX, the basin is approximately 5 miles wide.

The main tributaries of the York River at CAX are King Creek on the southeastern boundary of the station, and Cheatham Pond and Queens Creek on the northern boundary of the site. Drainage to Penniman Lake and Cheatham Pond is the main drainage feature of the station. Storm water runoff from the site enters into either the York River or the unnamed tributary to the York River.

Wetlands are found along all of the creeks that drain CAX and also along some shoreline areas of the York River. The tidal reaches of the York River extend throughout CAX. The tributary creeks draining CAX are also tidal up to 1 mile inland from the riverbank. The tidal reaches of the York River, including those in the vicinity of CAX, are classified as shellfish waters.

Site 1 is situated at the junction of the York River and a small, unnamed tributary to the York River. Surface water, as runoff at Site 1 would drain to the York River and unnamed tributary.

### **3.7 Ecology**

#### **3.7.1 Rare, Threatened, or Endangered Species{ TC "1.2.2 Rare, Threatened, or Endangered Species" \f C \l "3" }**

A review of Virginia Department of Conservation Natural Heritage Program (VDCR, 2001) and Virginia Department of Game and Inland Fisheries (VDGIF, 2001) county databases as well as the Cheatham Annex Natural Resources Management Plan (USDA, 1988) and Natural Heritage

Inventory (VDCR, 1990) indicated that no federally or state listed endangered or threatened species are currently using Cheatham Annex habitats. Suitable habitat exists on CAX for both the Red-cockaded woodpecker (federally endangered) and the Bald Eagle (federally threatened, state endangered), though neither species was resident during the 1992 Natural Heritage inventory. In the past, infrequent sightings of other endangered/threatened avian species, including the Peregrine falcon and Bachman's and Kirtland's warblers, were made in the general area (NEESA, 1984). Bordering the CAX property is the York River, which provides seasonal foraging habitat for federally and state endangered Kemp's Ridley sea turtles and federally threatened Loggerhead sea turtles (VIMS, 2001). The shoreline along the York River may also provide habitat for federally threatened Piping Plovers. Table 3-2 lists federal and state endangered, threatened, and sensitive species of concern known from York County.

Rare resources and communities identified from CAX in the Virginia Department of Conservation and Recreation (VDCR) Natural Heritage Program database and the CAX Natural Heritage Inventory include a significant great blue heron colony, low salt marsh and salt scrub habitats, coastal plain depression ponds, non-riverine wet hardwood forests, and coastal plain calcareous seepage swamps. The calcareous (marl) ravines and swamps were identified in the Natural Heritage Inventory as "state-significant rare plant habitats." VDCR also recognizes these areas as providing potential shell/marl groundwater habitat for the globally rare Tidewater amphipod (*Stygobromus araeus*) (Fleming, 2001). To protect these areas from invasion by exotic grasses and other threats, the Inventory recommended establishment of four special interest areas within the Cheatham Annex property line (VDCR, 1990). Site 1 is located outside the boundaries of the special interest areas but does include both low salt marsh and wetland hardwood forest habitats.

### **3.7.2 Habitats and Biota**

Table 3-3 summarizes an inventory of plant and animal species observed utilizing CAX during the 1988 United States Department of Agriculture (USDA) and 1990 VDCR surveys, the last two species surveys performed at the site. The following sections describe the terrestrial, aquatic, and wetland habitats of the general area and immediate vicinity of Site 1. { TC "1.2.3 Habitats and Biota" \f C \l "3" }

#### **3.7.2.1 Terrestrial Habitat**{ TC "1.2.3.1 Terrestrial Habitat" \f C \l "4" }

Terrestrial flora on CAX consists of predominantly woodland species. Three types of forest are present: pine stands composed primarily of loblolly and Virginia pines, mixed pine and hardwood stands, and hardwood stands. Elevated level areas are the predominant locations of pine stands, while hardwood stands are found on slopes and ravines. These wooded areas are important in reducing soil erosion and providing wildlife habitat. Native tree species found at CAX include american beech (*Fagus grandifolia*), black cherry (*Prunus serotina*), red maple (*Acer rubrum*), sweet gum (*Liquidambar styraciflua*), various pines, white ash (*Fraxinus americana*), and white oak (*Quercus alba*).

The woodland's understory is composed of various seedling trees and vine species, such as Virginia creeper, briars, and honeysuckle. Ferns are found in many moist, shaded areas. Ornamental trees and shrubs have been planted in the improved areas and along major roadways. None of the plant species that thrive at CAX are listed on the federal or Commonwealth endangered lists.

Small, undeveloped tracts of land at CAX support a variety of indigenous wildlife species. Whitetail deer (*Odocoileus virginianus*), beaver (*Castor Canadensis*), bobcats (*Lynx rufus*), red and gray foxes (*Vulpes vulpes* and *Urocyon cinereoargenteus*), squirrels (*Glaucomys volans*, *Sciurius carolinensis*, *Tamiasciurus hudsonicus*), raccoons (*Procyon lotor*), opossums (*Didelphis virginianus*), and rabbits (*Sylvilagus floridana*) are present. Game birds such as wild turkey (*Meleagris gallopavo*), quail, duck, and pheasants are also resident. Songbirds common to the eastern Virginia area are in abundance at CAX, along with a raptor population consisting of small hawks (*Accipiter striatus*, *Buteo jamaicensis*, *Buteo lineatus*, and *Falco spaverius spaverius*), owls (*Bubo virginianus*, *Otus asio*, and *Strix varia*), and osprey (*Pandion haliaetus carolinensis*). Carrion-feeding birds such as crows and turkey vultures (*Cathartes aura*) are also common. The southern bald eagle (*Haliaeetus leucocephalus leucocephalus*, listed on the federal threatened/state endangered lists) is known to nest nearby at WPNSTA Yorktown. Suitable habitat exists for roosting and perching in the area, but only occasional sightings of eagles have been made.

The Site 1 landfill surface was regraded and covered with two feet of soil in 1981. This area is now open, flat, and is overgrown with grassy vegetation. Dense mixed pine-hardwood woodlands that steeply descend to the unnamed creek bordering the site characterize the southern edge of the landfill where exposed landfill debris can be observed. Dominant canopy and understory tree species include tulip poplar (*Liriodendron tulipifera*), black walnut (*Jugland nigra*), american

beech (*Fagus grandifolia*), sweetgum (*Liquidambar styraciflua*), greenbriar (*Smilax rotundifolia*) and honeysuckle (*Lonicera carolinianum*). Osprey, snakes, and evidence of deer have been observed at the site.

#### 3.7.2.2 Wetlands/Aquatic Habitat

Site 1 is bordered to the north by the York River and to the west by the unnamed creek. The bank along the York River is extremely steep and is approximately 15 to 20 feet high. A narrow beach and broad tidal flat are present between the bank and the York River. The unnamed tributary bordering Site 1 is under tidal influence for part of its length and is flooded semidiurnally. Standing water can be found throughout the stream at low tide, providing significant foraging habitat for predatory picivorous birds and mammals. Frogs, insect larvae, small fish, water moccasins, and fiddler and mud crabs have been observed along the creek, providing a food source for upper trophic level receptors. Salinities in this area of the York River estuary can be characterized as mesohaline (from 15 to 20 parts per thousand (ppt)), and can fluctuate depending on seasonal impacts, runoff, and rainfall.

Of the 295 fish species known from the Chesapeake Bay, only 32 are year-round residents (CBP, 2001). Nursery areas, foraging areas, and spawning ground attract the remaining species from the Atlantic Ocean and freshwater tributaries each year. In the York River, resident fish include hogchoker (*Trinectes maculatus*), weakfish (*Cynoscion regalis*), and oyster toadfish (*Opsanus tau*). Spot (*Leiostomus xanthurus*), and croaker (*Micropogonias undulatus*) are common in nursery and foraging areas in the summer and numerous anadromous and catadromous fish utilize the area during migration, including the alewife (*Alosa pseudoharengus*), american eel (*Anguilla rostrata*), american shad (*Alosa sapidissima*), blueback herring (*Alosa aestivalis*), striped bass (*Morone saxatilis*), and white perch (*Morone americana*) (NOAA, 1993). Commercially and recreationally important species from the York River include american shad, bay anchovy (*Anchoa mitchilli*), blue crab (*Callinectes sapidus*), bluefish (*Pomatomus saltatrix*), croaker, spot, striped bass, summer flounder (*Paralichthys dentatus*), and weakfish (NOAA, 1994). The York River in the vicinity of CAX is a designated crab pot fishery from March through November of each year; immediately north of CAX is a spawning and nursery ground for blue crabs. Several species of endangered sea turtles (namely the green (*Chelonia mydas*), hawksbill (*Eretmochelys imbricata*), leatherback (*Dermochelys coriacea*), loggerhead (*Caretta caretta caretta*), and Kemp's Ridley (*Lepidochelys kempii*)) are known to feed in the Chesapeake Bay and

occasionally forage in the York River, including in the vicinity of CAX during the summer (VIMS, 2001).

The York River is designated as Essential Fish Habitat (EFH) for three species of fish managed by the Mid-Atlantic Fishery Management Council: summer flounder (*Paralichthys dentatus*), bluefish (*Pomatomus saltatrix*) and butterfish (*Peprilus triacanthus*). Though both the bluefish and butterfish use the more open, pelagic waters characteristic of the river, juvenile summer flounder often utilize unvegetated nearshore sandy-bottoms and salt marsh creeks similar to the unnamed tributary habitat at Site 1 as nursery areas (MAFMC, 1998). Other species likely to use salt marsh creeks include anchovies (*Anchoa mitchelli*), blue crabs, juveniles of migratory species, hard and soft-shell clams (*Mercenaria mercenaria* and *Mya arenaria*), killifish, minnows, and mummichogs (*Fundulus* and *Cyprinodon* spp.), oysters (*Crassostrea virginica*), silversides (*Menedia* spp.), and weakfish (NOAA, 1993).

Wetlands are mainly found along principal tributaries to the York River and along the York River shoreline at CAX. Four major marsh types exist in the vicinity: salt-marsh cordgrass communities, big cordgrass communities, cattail communities, and brackish water mixed communities. Wetlands are often grouped into classifications based on their estimated environmental value per acre and provide primary habitats for migrating waterfowl and nursery habitat for resident and transitory fish species. Salt-marsh cordgrass and brackish water mixed communities are characterized by the highest productivity and use by waterfowl and wildlife, and maintain close associations with fish spawning and nursery areas. They are also important to the shellfish industry and as shoreline erosion inhibitors. The majority of wetlands on CAX are of this type. Big cordgrass and cattail community type marshes are found at higher elevations, where there is less opportunity for detritus (loose soil or organic particles) to be washed into nearby waterways by the tides. These communities are valuable as flood buffers.

One wetland area was identified at Site 1. This wetland was identified as an estuarine intertidal persistent emergent wetland, is associated with the unnamed tributary bordering the site, and is approximately 1.25 acres in size. This determination was based on a wetlands inventory conducted at Site 1 in early 2000 with a field verification conducted on March 27, 2000. Figure 3-3 shows the location of the wetlands with respect to Site 1. A break exists in wetland vegetation types within the wetland where the estuarine tidal influence from the York River ends. At this point the vegetation changes from herbaceous, saltmarsh vegetation dominated by the common reed and carolina sea-lavender to more scrub/shrub vegetation dominated by bald cypress, southern bayberry, and water horsetail. Further upstream the wetland habitat is

characterized by more forested vegetation types, including sycamore, swamp cotton-wood, and southern bayberry. The wetland habitat is described in more detail in the wetland delineation report in Appendix A.

#### **4.0 REMEDIAL INVESTIGATION FIELD ACTIVITIES**

The purpose of this section is to summarize the RI field investigations that were implemented at Site 1. The RI investigations for Site 1 were conducted in November 1999, May 2000, August 2001 and November 2001 and consisted of a soil, groundwater, surface water and sediment investigations. Data from other investigations have been combined where appropriate with the RI to provide a more comprehensive data set. General sampling information such as numbers and locations of these non-RI samples will be discussed herein. However, sampling procedures are not included in this report. These details may be found elsewhere. Specifically, refer to the following reports:

- Surface soil and subsurface sampling - Weston SI Report (Weston, 1994)
- Surface water and sediment sampling - Baker Field Investigation Report (Baker, 1999)
- Toxicity Characteristic Leaching Procedure (TCLP) sampling - Baker Construction Closeout Report (Baker, 2000)

The physical results/findings of this investigation are also presented in this section. Analytical results as well as nature and extent are discussed in Section 5.0.

A summary of the samples collected during RI and non-RI investigations is presented in Table 4-1. It should be noted that not all data from the non-RI investigations was used, particularly groundwater. Groundwater samples were collected in 1992 and 1997, however this data was not used. Only groundwater samples collected in 2000 will be evaluated in this RI for several reasons. As mentioned in Section 2.2 some of the monitoring wells were constructed with inappropriate materials or inadequately located. Additionally, different purge and sampling methods employed beginning in 1997 render historical trend analysis meaningless.

#### **4.1 Non-RI Investigations**

The 1992 data combined in this RI data set include two soil borings. These borings were advanced using conventional means (i.e., auger rig and split-spoon sampling). Samples were collected for laboratory analysis from three intervals; near surface (0.5 feet bgs), immediately above the groundwater table, and between the near-surface and groundwater table. Samples were

analyzed for VOCs, SVOCs (acids and base/neutrals), total organic carbon (TOC), TPH, PCBs and dioxins, and total metals. During the 1992 SI two soil boring samples, 1-SB07-3A and 1-SB08-3A, were collected from ash within the landfill and analyzed for dioxins. Dioxins were not detected in either sample. The detection limit was 0.3 ng/g. Therefore, dioxins were not analyzed in subsequent sampling events.

The 1998 data combined in this RI data set include two soil borings as well as three surface water and four sediment samples. These borings were advanced via hand auger, with one surface soil sample (0 to 6-inches bgs) and one shallow subsurface soil sample (12 to 24-inches bgs) collected from each location. Sample locations are shown on Figure 4-1, and include 1-HA01 and 1-HA02. The surface water and sediment samples were collected from the marsh adjacent to (and northwest of) the landfill that is associated with the unnamed tributary to the York River. Samples were analyzed for TCL VOCs and SVOCs, TAL metals and cyanide, and nitramine compounds.

#### **4.2 1999 RI Investigation**

The 1999 RI investigation included soil borings and soil sampling. Sample locations are shown on Figure 4-1. This field investigation included:

- Ten direct-push borings that were advanced through the landfill (1-DPB01 through 1-DPB10)
- Five hand auger borings (99-EXP01 through 99-EXP05)
- Five verification borings related to the "DPB" series borings

The ten direct-push borings were advanced in the landfill on November 15 and 16, 1999 via Geoprobe rig with a Macro-Core Sampler (Photographs 1 and 2). The purpose of the boring program was to characterize the landfill and cover material and delineate the extent of the landfilled materials. The Macro-Core Sampler can collect soil samples up to 4-feet in length by 2-inches in diameter. Test Boring Records for these borings are included in Appendix B. To collect a sample, a clean, virgin acetate liner was inserted into a decontaminated Sampler (see Section 4.6) and the Sampler assembled and attached to drive rods. A Geoprobe hammer was then used to drive the Sampler and rods to the desired depth. The Sampler was then removed

from the ground, disassembled, and the acetate liner removed. The liner was cut open using a decontaminated knife and screened with a photo ionization detector (PID) (Photograph 3). Locations within the soil core to be sampled for laboratory analysis were based on PID readings and material observed in the core. Samples for VOC analysis were placed directly into a sample container with minimal disturbance using a decontaminated stainless steel sampling spoon. The remaining portions were thoroughly homogenized in a clean, virgin aluminum pie pan and then placed into appropriate containers (Photograph 4). Soil samples collected from these borings were submitted for laboratory analysis of TCL organics, TAL inorganics and cyanide, and nitramine compounds.

The five hand auger borings were advanced in the landfill on November 16, 1999. The purpose of these borings was to help delineate of the extent of the landfilled materials. No samples for laboratory analysis were collected. A stainless-steel bucket auger was advanced by hand until native soils were encountered. Stainless-steel bucket augers and rods were decontaminated according to the procedures outlined in Section 4.6.

One verification boring was located next to each of the following borings: 1-DPB02, 1-DPB04, 1-DPB08, and 1-DPB10. Macro-Core Sampler recovery was poor for some samples at each of the borings. The purpose of the verification borings was to collect soil samples via conventional drilling methods to fill lithology data gaps created by poor sample recovery. Verification borings were advanced using a truck-mounted drill rig that employed stainless-steel split spoons without augers. Subsurface soil samples were collected with a split-spoon sampler in accordance with American Society for Testing and Materials (ASTM) Method D 1586-84 (ASTM, 1984) (Photographs 5 and 6). Stainless steel split-spoon samplers were decontaminated prior to sample collection according to the procedures outlined in Section 4.6.

### **4.3 2000 RI Investigation**

The 2000 RI investigation included soil borings and sampling, monitoring well installation and groundwater sampling. Sample locations are shown on Figure 4-1. This field investigation included:

- Two groundwater monitoring wells installed downgradient of the landfill (1-GW09 and 1-GW10)

- Six Punch Borings (00-PB01 through 00-PB06)
- Two soil samples for geotechnical analysis (1-GEO-01 and 1-GEO-02)
- Five groundwater samples from wells 1-GW05, 1-GW06, 1-GW07, 1-GW09, and 1-GW10

#### **4.3.1 Monitoring Well Soil Borings**

Two monitoring well soil borings were advanced to determine stratigraphy, accommodate subsurface soil sampling, and facilitate monitoring well installation. The borings also allowed for identification of hydrogeologic units and general assessment of hydrogeologic conditions. The borings were advanced and wells installed and developed on April 26 and 27, 2000. Monitoring well installation is discussed in Section 4.3.2.

4-1/4 inch inside diameter (ID) hollow stem augers were used to advance the soil borings. Split- spoon samples were collected through the inside of the augers per ASTM Method D 1586-84 (ASTM, 1984). Split-spoon samples were collected at continuous 2-foot intervals from the ground surface until borehole completion. Each split-spoon sample was classified visually by the field geologist. The classification included characterization of soil type, color, moisture content, consistency, and other pertinent information such as evidence of contamination. Lithology descriptions are provided on the Test Boring and Well Construction Records in Appendix B. Samples were not collected for laboratory analysis.

#### **4.3.2 Monitoring Well Installation**

Two permanent monitoring wells (1-GW09 and 1-GW10) were installed and developed at Site 1 at the locations shown on Figure 4-1. The monitoring wells were installed to facilitate collection of groundwater samples for characterizing conditions in the shallow water-bearing zone, and for determining groundwater elevations. Both wells were installed hydraulically downgradient of the landfill to provide additional characterization of groundwater conditions in that location.

Both shallow monitoring wells were installed upon advancing borings to the desired depth via hollow stem augering as described in Section 4.3.1. Well 1-GW09 was installed to approximately 28 feet bgs, and 1-GW10 to approximately 18 feet bgs.

Construction data for the monitoring wells are summarized in Table 4-2 and individual well construction diagrams are shown on the Test Boring and Well Construction Records provided in Appendix B. The wells were constructed of 2-inch nominal diameter, Schedule 40 flush-joint and threaded polyvinyl chloride (PVC). The screened interval for each well was 15 feet and consisted of 0.01-inch slotted screen. A fine quartz sand pack (No. 1 sand), was placed in the annulus between the screen and the auger walls extending at least 2.0 feet above the top of the screen. A bentonite pellet seal of 2.0 feet minimum thickness was then placed above the sand pack and hydrated with potable water. As the sand and bentonite pellets were placed through the augers, the augers were slowly pulled from the ground to allow the screen pack and seal material to settle in the boring annulus. The pellets hydrated for a minimum of 45 minutes prior to resumption of well construction activities. The remaining annular space was backfilled with a cement-bentonite grout. The PVC riser extended approximately 2.5 feet above ground surface. This "stick-up" section is protected by a 4-inch square, 5-foot long steel casing (with locking cap) that was set in the cement-bentonite grout. Concrete pads 3-feet by 3-feet by 6-inches thick were constructed around each well. Three bollards (concrete-filled posts) were set around the well within the concrete pad. The posts were set to a minimum depth of 2.5 feet bgs and painted orange. Each monitoring well was labeled with a well tag that was engraved with general well construction information indicating the well is not for consumptive use.

Following well construction and curing of the grout seals (i.e., typically 24 hours or more), each well was developed to remove fine-grained sediment and to establish an interconnection between the well and the formation. Prior to commencement of development, water level and total depth were measured to determine the water column within each well. The volume of water in the well was calculated from the height of the water column. The monitoring wells were developed by pumping water and fine-grained sediment from the wells using a pneumatic (double diaphragm) pump. Thirteen well volumes were evacuated from each well.

Field measurements of pH, specific conductance, oxidation-reduction potential (Eh), and temperature were made prior to development and after each well volume was removed to provide data regarding groundwater quality and well stabilization. These measurements were recorded in a field logbook and are presented on Table 4-3. Development was considered complete when the final three field measurements were each within ten percent of the preceding measurement or after a pre-determined maximum well volume was deemed sufficient by the Baker Field Geologist. Development water was handled in a manner specified in Section 4.7.

### **4.3.3 Punch Borings**

The six punch borings were advanced by continuously driving split-spoon samplers to the desired depth. Work for the punch borings was performed on April 27, 2000. Split-spoon samples were collected through the inside of the augers per ASTM Method D 1586-84 (ASTM, 1984). Split-spoon samples were collected at continuous 2-foot intervals from the ground surface until borehole completion. Each split-spoon sample was visually classified by the field geologist. The classification included characterization of soil type, color, moisture content, consistency, and other pertinent information such as evidence of contamination. Lithology descriptions are provided on the Test Boring and Well Construction Records in Appendix B. The purpose of these borings was to assist with delineation of landfill extent, so samples were not collected for laboratory analysis.

### **4.3.4 Geotechnical Samples**

Two soil samples were collected from the bank of the York River tidal flat for geotechnical analysis on April 27, 2000 (Figure 4-1). Geotechnical samples could not be obtained from 1-GW09 due to volume requirements. The sample location was near 1-GW09 and taken from the bank at 4-6 feet and 8-10 feet below the top of the bank. These levels corresponded to specific lithology identified in the boring for 1-GW09 based on elevation and lithology characteristics. Soil samples were collected from the bank using a steel spade shovel and placed directly into two 5-gallon plastic buckets, one for each sample. Decontamination was not required as the samples were not collected from waste materials or analyzed for chemical constituents. The samples were analyzed for one point standard proctor, moisture content, grain size analysis, and direct shear.

### **4.3.5 Hazardous Waste Characteristics Sampling**

A total of five waste material samples were collected in various locations. The purpose of this sampling was not to characterize the entire landfill as hazardous or non-hazardous, but to identify the potential of specific materials to be hazardous.

Three samples of beach area debris were collected in August 1999. Sample 01-TCRA01 consisted of exposed debris, sample 01-TCRA02 was of the soil matrix from a debris zone, and sample 01-TCRA03 was a dried yellow residue from a rusted metal bucket (Baker 2000). Two of

three samples were analyzed for hazardous waste characteristics, including metals in the TCLP leachate, reactivity, corrosivity, and ignitability. Sample 01-TCRA03 was analyzed for only lead in the TCLP leachate. Two composite samples of landfilled materials (primarily ash) were collected in November 1999. Sample 1-DPB04-COMP is a composite sample of ash material observed in samples from boring DPB04. Sample 1-DPBCOMP-01 was comprised of one aliquot of predominantly ash material from each boring. Both samples were analyzed for hazardous waste characteristics, including VOCs, SVOCs, and metals in the TCLP leachate, reactivity, corrosivity, and ignitability.

#### **4.3.6 Groundwater Sampling**

Five groundwater samples were collected on May 15, 2000, from wells 1-GW05, 1-GW06, 1-GW07, 1-GW09, and 1-GW10. Groundwater sampling procedures included three primary procedures:

- Depth to water and well depth measurements
- Groundwater purge
- Groundwater sample collection

Prior to groundwater purging, a water level measurement from each well was obtained. The total well depth was also recorded from each well to the nearest 0.01-foot prior to sampling. Water level and well depth measurements were used to calculate the volume of water to be purged from each well.

Groundwater present in the well and surrounding sand pack is not typically representative of aquifer groundwater conditions. This water must be purged prior to sample collection to ensure that representative groundwater is sampled. A low-flow purging technique was employed in groundwater sample acquisition. The low flow purging entails pumping a well at a flow rate below the recharge capacity of the formation. A flow rate of approximately 0.5 liters per minute was used for this sampling event. As a determination that representative groundwater samples are being collected, the change in certain water quality parameters (WQPs) is monitored. The WQPs for Site 1 included pH, specific conductance, dissolved oxygen, temperature, and turbidity, and were recorded at each well volume. Standard practice for determining that representative groundwater is being sampled is observance of stabilization in the WQPs. The criteria for determining stabilization of WQPs (for three successive readings) included:

- pH  $\nabla$  0.1 standard units
- Specific conductance  $\nabla$  10%
- Turbidity  $\nabla$  10%
- Dissolved oxygen  $\nabla$  10%

A peristaltic pump (GeoPump7) was used to purge and sample with the intake set three feet into the static water column. Virgin polyethylene tubing and silicon pump-head tubing were used to convey groundwater. WQP stabilization occurred within three to five well volumes.

Groundwater sampling documentation included specific sample information such as well number, sample identification, time and date of sample collection, sampling team, WQP measurements, and analytical parameters. These items were recorded in a field logbook and on the sample labels. Purge information is presented in Table 4-4.

Upon WQP stabilization, groundwater samples were collected. Samples were collected in order of relative volatility. The sample for VOCs was collected first, followed by the SVOC sample, then pesticides/PCBs, and finally total and dissolved metals.

#### **4.4 2001 Additional Sampling for RI**

The 2001 RI investigation included soil borings and sampling, surface soil sampling, surface water and sediment sampling. Sample locations are shown on Figure 4-1. This field investigation included:

- Ten surface soil samples, and two duplicate samples (CX01-SS01 through CX01-SS10-00)
- Four subsurface samples, and one duplicate sample (CX01-SB01 through CX01-SB04)
- Four surface water samples, and one duplicate sample (CX01-SW01 through CX01-SW04)
- Twenty-two sediment samples, and two duplicate samples, (CX01-SD01-01 through CX01-SD11-01 and CX01-SD01-02 through CX01-SD11-02)

#### **4.4.1 Soil Investigation**

Surface soil samples were collected at 10 locations as shown on Figure 4-1. Four of the sample locations were outside of the landfill area, within the drainage swale southwest of the landfill. Three of the samples were collected within the debris area to the north of the landfill boundary. The other three samples were collected west of landfill area boundary. These additional samples were collected for ecological risk screening purposes and are discussed in Section 8.0 of this report. Surface soil samples were collected from the 0 to 6 inch interval. Each sample was analyzed for Target Compound List (TCL) VOCs, SVOCs, pesticides, PCBs, Target Analyte List (TAL) metals and cyanide.

Subsurface soil samples were collected at four locations adjacent to the landfill (drainage swale southwest of the landfill) from the 6 to 24 inch interval at each location. Each sample was analyzed for Target Compound List (TCL) VOCs, SVOCs, pesticides, PCBs, Target Analyte List (TAL) metals and cyanide. Sample locations are provided on Figure 4-1.

Additionally, all soil sampling locations were surveyed via the utilization of a Global Positioning System (GPS). All surveyed locations were established to the State Plane Coordinate System, Virginia South, Datum (1983) with units in feet.

#### **4.4.2 Surface Water Investigation**

Four surface water samples were collected as part of this investigation. Two were collected from locations within the stream west of the landfill. The stream, an unnamed tributary to the York River, flows through a wetland. One sample, CX01-SD04 was collected outside of the landfill area to the north (within the above referenced wetland near the metal storage tank located at the top of the landfill on the northwestern edge) from a depositional pool on a bend in the stream. Another surface water samples, CX01-SW01 was collected outside of the landfill area from a stagnant pool of water directly under a culvert. The surface water samples are used in the ecological risk assessment (see Section 8.0) Each surface water sample was analyzed for TCL VOCs, SVOCs, pesticides, PCBs, TAL total and dissolved metals and cyanide. In addition, field measurements for pH, temperature, salinity, dissolved oxygen, and conductivity were recorded at these surface water locations as shown on Table 4-5. Sample locations are provided on Figure 4-1.

Additionally, all surface water samples were collected during the low tide cycle to avoid the possible interference of the York River.

#### **4.4.3 Sediment Investigation**

Sediment samples were collected from eleven locations within the stream west of the landfill as shown on Figure 4-1. Nine of the samples were collected along the unnamed along the western border of the landfill. One sample, CX01-SD04 was collected outside of the landfill area to the north (within the above referenced wetland near the metal storage tank located at the top of the landfill on the northwestern edge) from a depositional pool on a bend in the stream. Sample CX01-SD01 was collected outside of the landfill area under a culvert, stagnant water was present at this location. At each location both a surface sediment (0 to 4 inches) and subsurface sediment (4 to 8 inches) sample was collected. The samples were collected for use in an ecological risk assessment (Section 8.0). Each sediment samples was analyzed for TCL VOCs, SVOCs, pesticides, PCBs, TAL metals and cyanide.

All sediment samples were collected during the low tide cycle to avoid the possible interference of the York River.

#### **4.5 Quality Assurance/Quality Control (QA/QC) Samples**

Four types of field QA/QC samples were collected and analyzed including duplicate samples, trip blanks, equipment rinsate blanks, and field blanks. A summary of QA/QC samples is presented on Table 4-1. The QA/QC sample types collected are defined below:

- **Duplicate Sample**: Two samples collected simultaneously and placed into separate containers from the same source under identical conditions. Field duplicates are primarily used to check the precision and consistency of the sampling procedures used. The field duplicate also serves as a check on the analytical procedures. One duplicate sample is typically collected for every 10 environmental samples collected (10 percent frequency) for each media type (surface soil, subsurface soil, and groundwater).
- **Trip Blanks**: Samples that originate from analyte-free water taken from the laboratory to the sampling site and returned to the laboratory with the VOA samples. One trip blank accompanies each cooler that contains VOA samples. Laboratory analysis of the trip

blank ensures that the sample containers were not contaminated prior to receipt or during transport to the laboratory.

- Equipment Rinsate Blank: Samples obtained by pouring laboratory supplied deionized water over/through sample collection equipment after it has been decontaminated. These samples are used to determine if decontamination procedures are adequate.
- Field Blank: Samples obtained from each water source utilized during the field program. Analysis of field blanks will indicate whether contamination was introduced into the samples during the collection process.

#### **4.5.1 1992 QA/QC Samples**

The 1992 investigation QA/QC sampling included one trip blank and one rinsate blank related to the Site 1 investigation. The rinsate blank was collected from soil sampling equipment. Duplicate samples were collected in the overall investigation that included three sites. None were collected at Site 1.

#### **4.5.2 1998 QA/QC Samples**

The 1998 investigation QA/QC sampling included two trip blanks, one field rinsate blank, and two equipment rinsate blank samples associated with soil sampling. Rinsate blank sample 388-RS01 was taken from a sampling spoon, and 388-RS02 from a hand auger. Soil duplicates were collected at a frequency of 10%, with two collected at Site 1.

#### **4.5.3 1999 QA/QC Samples**

The 1999 investigation QA/QC sampling included one trip blank, two field rinsate blanks, and three equipment rinsate blanks. Rinsate blank sample 104-RS01 was taken from a sampling spoon, 104-RS03 from a mixing pan, and 104-RS04 from a hand auger. Soil duplicates were collected at a frequency of 10%, with one collected at Site 1.

#### **4.5.4 2000 QA/QC Samples**

The 2000 investigation QA/QC sampling included one trip blank, one field rinsate blank, and one equipment rinsate blank. It should be noted that the field and equipment rinsate blanks were collected at Site 1, but for a pond-sampling event. Both blank samples are considered to be representative of groundwater sampling conditions. The field blank sample exhibited no detections of organic compounds and low detections of common inorganic constituents, such as calcium and sodium. This is generally consistent with historical field blanks. The equipment rinsate blank was taken from polyethylene tubing; the same type used for groundwater sampling. One groundwater duplicate was collected at Site 1.

#### **4.5.5 2001 QA/QC Samples**

The 2001 investigation QA/QC sampling included four trip blanks and three equipment rinsate blanks collected during the August 2001 sampling event. Rinsate blank sample CX01-RS01 was taken from a stainless steel spoon, CX01-RS02 from an acetate sleeve and CX01-RS03 from an aluminum pie pan. One equipment rinsate, 1116RB, taken from a stainless steel spoon and one trip blank were collected in the November sampling event. Soil duplicates were collected at a frequency of 10%, with two collected at Site 1.

### **4.6 Sample Handling and Analysis**

The following sections present information regarding sample preservation, handling, and chain-of-custody procedures.

#### **4.6.1 Sample Preservation, Handling, and Analysis**

Analytical methods and laboratory turnaround times are presented in the Project Plans for Field Investigations, Engineering and Environmental Support (Sites 1, 4, 7, 11, and AOCs 1 and 2), November 3, 1999. Preservation requirements, bottle requirements and holding times were in accordance with the specific analytical method. Field activities related to sample preservation and handling were conducted according to Baker's Standard Operating Procedures (SOP) F301 and USEPA Region III protocol.

#### **4.6.2 Chain-of-Custody Forms**

Chain-of-custody procedures were followed throughout the field program using Baker SOP F302 to ensure a documented, traceable link between analytical results and the sample/parameter that they represent. These procedures are intended to provide a legally acceptable record of sample collection, identification, preparation, storage, shipping, and analysis.

A chain-of-custody form was completed for each shipping container in which the samples were shipped. After the samples were properly packaged, the containers were sealed and prepared for shipment. Custody seals were placed on the outside of the containers to ensure that the samples were not disturbed prior to reaching the laboratory. Chain-of-Custody forms for the remedial investigations are presented in Appendix C.

#### **4.7 Decontamination Procedures**

Decontamination procedures performed in the field were conducted in accordance with USEPA Region III guidelines. In general, sampling and drilling equipment was divided into two decontamination groups: heavy equipment and routine sample collection equipment. Heavy equipment included the drill rig, hollow-stem augers, and drill rods. Routine sample collection equipment included split-spoons and stainless-steel sampling spoons.

For heavy equipment, the following decontamination procedures were implemented:

- Remove caked-on soil with brush as necessary
- Steam clean with high-pressure steam
- Air dry

For routine sample collection equipment, the following decontamination procedures were implemented:

- Wash equipment thoroughly with laboratory detergent (Liquinox7) and potable water using a brush to remove particulate matter or surface film

- Rinse thoroughly with distilled water
- Rinse with 10 percent nitric acid solution
- Rinse thoroughly with distilled water
- Rinse with isopropanol
- Air dry
- Wrap in aluminum foil if not to be used immediately

Decontamination fluids generated for these procedures were managed according to the procedures outlined in Section 4.7.

#### **4.8 Investigation Derived Waste Management**

The following types of Investigation Derived Waste (IDW) were generated during the field investigation:

- Excess soil from direct-push operations (1999)
- Excess soil from monitoring well installation (2000)
- Decontamination fluids from steam cleaning heavy equipment and from routine sampling equipment and well development and purge water (1999 and 2000)

Excess soil from the 1999 direct-push operations was temporarily stored in one 55-gallon drum and staged at the site, to approximately ¼ full. This soil was combined with waste material collected from the beach during the TCRA conducted in early 2000. The TCRA material and direct-push soils were contained in five 55-gallon drums, which has since been disposed. Appendix D contains IDW disposal documentation.

Excess soil from 2000 monitoring well installation was temporarily stored in approximately three 55-gallon drums and staged at the site. This soil is non-hazardous and will be disposed of onsite, outside the landfill area.

IDW liquids include decontamination fluids from steam cleaning heavy equipment and routine sampling equipment, as well as well development and purge water. Liquid IDW generated during the 1999 investigation was combined with liquid IDW generated from other sites investigated at Cheatham Annex and disposed. Approximately 250 total gallons were generated. Appendix D contains IDW disposal documentation. Liquid IDW generated in 2000 included heavy equipment and well development and purge water. This IDW was collected and temporarily stored in approximately two 55-gallon drums and staged at the site. The IDW liquid was disposed as non-hazardous waste by IMS in July 2000 at an appropriate facility.

#### **4.9 Surveying**

All monitoring wells and borings advanced prior to 1999 were surveyed by a registered surveyor (Patton, Harris, Rust & Associates, P.C.) to determine elevation (vertical control) and location (horizontal control). At each well, the top of PVC casing, and ground surface were surveyed. Coordinates for the monitoring wells are presented on the well construction records in Appendix B. Top of PVC casing and ground surface elevations are presented on Table 4-2. Baker personal located 1999 and 2000 borings and monitoring wells based on a point with known location and elevation. Horizontal positions are based on the North American Datum (NAD) 83 Datum, Virginia State Plane, South Zone. Elevations are based on the North American Vertical Datum (NAVD) 88 Datum.

#### **4.10 Site-Specific Hydrogeology and Extent of Landfill**

Site 1 can be thought of as two separate environments; landfilled materials and native soils. The landfill itself is comprised of three areas, including landfilled materials, impacted soils, and surface debris piles. This section describes the findings of the RI relative to the extent of the landfill and the character of the surrounding hydrogeology.

Figure 4-2 shows the current areal extent of the landfill, as well as the different areas of the landfill. Figure 4-2 also shows current topography and the original topography (prior to the landfill). Vertical cross-sections A-A' and B-B' through the landfill and surrounding area are

shown on Figure 4-3. Section 2.0 provides a detailed discussion of the landfill contents, which will not be reiterated here. Section 2.0 also provides a landfill volume estimate. This estimate was developed from information provided by the borings advanced in and around the landfill. A computer program called "Soft Desk" was used to estimate the volume of landfilled materials. This was done by creating model surfaces that represent the current ground surface and original ground surface based on boring information. These surfaces were used to create a three-dimensional model of the landfilled materials, from which volume calculations were made.

The extent of buried ash and other wastes is shown on Figure 4-2. This extent is approximately 1 acre in area and is comprised of approximately 13,650 cubic yards of material. The area of impacted soils is approximately 0.3 acres and is comprised of approximately 1,100 yards of soil and debris. The cross sections on Figure 4-3 show that the landfill is predominantly ash, with a variable thickness. These cross sections also show that the landfill was built up and out from the original bank to the York River and tributary.

In November 2001, Baker was directed based on conversations with LANTDIV and Naval Weapons Station Yorktown (WPNSTA) representatives to perform trenching activities at Site 1. Trenches at Site 1 were advanced in order to obtain additional information as to the nature of the Landfill contents and the extent (both horizontal and vertical) of the buried materials

Based on trenching operations, previous soil borings, and monitoring well installations the estimated horizontal and vertical extent of the landfill has been determined at Site 1. Two volume estimates were generated for Site 1. This waste thickness estimate was contoured based on available data from trenching, borings, hand auger samples and monitoring well installations collected over the last four years. An estimate of the landfill cover thickness and its corresponding volume was obtained by using the same information as identified above. This cover estimate was also contoured. Thickness of the landfill waste varied from inches to greater than 19.5 feet. Thickness of the landfill cover varied from inches to 3.5 feet. The deepest location of the landfill was found to be near the northwestern portion of the site, adjacent to the fence line corner.

Additionally, another deep area was encountered in the southwestern portion of the site and appears to travel in an easterly direction, away from the southwest fence line corner. This potentially indicates that Site 1 has two distinct deep fill areas located within the Landfill.

The waste stream comprising the majority of the landfill contents consisted of black ashy soil containing broken glass, slag, cinders, concrete, brick, wood debris, asphalt and metal. Size of metal debris ranged from less than one inch to several feet in diameter. An area of large debris consisting of junk cars and helicopter parts is located immediately north of the landfill and was not included as part of the landfill delineation. However, volume of this debris pile is estimated at 1,100 cubic yards (cy).

Volume of the landfill waste contents at Site 1 is estimated to be approximately 12,500 cy. It should be noted that this number was obtained by taking the total volume generated and adding a conservative factor of 10 percent to account for any data gaps that possibly exist at the landfill.

Volume of the landfill cover at Site 1 is estimated to be approximately 3,900 cy. This number was also obtained in the same manner as the waste. A conservative factor of 10 percent was added to the volume of the cover to account for any possible data gaps.

A total volume of landfill waste contents and cover materials removal is approximately 16,400 cy.

The landfilled material was laid upon a native soil bank. Based on the regional geology, these uppermost soils are unconsolidated sediments of the Windsor Formation (Table 3-1). The Windsor Formation at Site 1 was observed to be comprised mainly of fine sand and silt. Thin, discontinuous clay layers are also present. The thickness of the Windsor Formation sediments at the site varies considerably (from less than 5-feet to greater than 20-feet). This is due to historical erosion of the York River. Immediately below the Windsor Formation is the Yorktown Formation. This formation is distinguished by the presence of shell fragments. At Site 1, the Yorktown Formation was observed to be comprised mainly of fine sand with lesser to nearly equal amounts of shell fragments. The Yorktown Formation was observed between 6-feet bgs at well 1-GW10 (Figure 4-3) and 25-feet bgs at well 1-GW08 (also Figure 4-3).

The aquifer observed in the vicinity of Site 1 appears to be the Yorktown-Eastover aquifer. Based on the groundwater study at WPNSTA Yorktown (USGS, 1997), this aquifer occurs as the water table aquifer and is unconfined in areas close to the York River and major tributaries.

In May 2000, the depth to groundwater at Site 1 ranged from approximately 9 feet below top of casing (at 1-GW10) to approximately 13 feet below top of casing (at 1-GW05). The depth to groundwater has varied over time (for examples, see Weston, 1994 and Baker, 1999). The depth

to groundwater varies spatially due to topography, and temporally due to the seasonal variability of rainfall and evapotranspiration. Despite these temporal variations, groundwater flow direction at Site 1 has historically been toward the York River. It has been observed at WNPSTA Yorktown that tides in the York River estuary induce near-shore fluctuations in groundwater levels, and that response diminishes with distance inland (USGS, 1997). This is a likely phenomenon at Site 1 given its close proximity to the York River.

Figure 4-4 shows a groundwater flow composite for two time periods, October 1998 and May 2000. Groundwater flow is toward the York River at both periods. This groundwater flow trend has been historically consistent and observed in previous investigations (Weston, 1994). The estimated hydraulic conductivity in the Columbia and Cornwallis Cave aquifers is similar at Site 1. The Hydraulic conductivity in the Columbia aquifer ranges between 0.4 feet/day to 8 feet/day, with an average of 2.45 feet/day (USGS, 1997). Hydraulic conductivity in the Cornwallis Cave aquifer ranges from 0.3 feet/day to 9 feet/day, with an average of 4.62 feet/day (USGS, 1997).

## **5.0 NATURE AND EXTENT OF CONTAMINATION**

This section presents the results of the RI performed at Site 1. The objectives of this section are to characterize the nature and extent of contamination. This characterization was accomplished by specific laboratory analysis of environmental samples including soil, groundwater, surface water, and sediment. Data summary tables and detection figures referenced in the text are presented at the end of this section.

Presentation of the analytical data includes a comparison of site data to established standards and/or criteria (Appendix E). The standards and criteria chosen for evaluation are media specific and help to provide a reasonable assessment of site conditions. An explanation of each of the standards and criteria is presented in Section 5.4.

### **5.1 Data Quality**

The quality of the data collected as part of the field investigations has been assessed by its accuracy and precision with respect to prescribed requirements or specifications for laboratory analysis. To make these determinations, an independent third-party validator performed data quality evaluations. Data were evaluated in accordance with the criteria established by USEPA guidelines, Laboratory Data Validation Functional Guidelines for Evaluating Organic Analyses (USEPA, 1991a) specific method requirements in OLM02.1, USEPA Region III modifications, Level D data requirements, and professional judgement. Validation of the analytical data serves to reduce the inherent uncertainties with its usability.

Additionally, analytical data were evaluated to determine both the usability of results, as well as contractual compliance relative to deliverables and the aforementioned requirements. Data validation also provided an interpretation of the reported quality control results. A minimum of ten percent of all laboratory calculations were verified as part of this validation. In addition, all instrument output (i.e., spectra, chromatograms, etc.) for each sample was carefully reviewed. Data quality was evaluated based on, but not limited to the following criteria:

- Data completeness
- Holding times
- Calibrations
- Blanks

- Surrogate recoveries
- Laboratory control samples
- Laboratory and field duplicates
- Matrix Spike/Matrix Spike Duplicate (MS/MSD) samples
- Internal standard performance
- Compound identification
- Compound quantitation

Based upon the results of this evaluation, various qualifiers and/or codes have been attached to certain data by either the laboratory or by the data validator with regard to the data's usability. These qualifiers often pertain to QA/QC problems and generally indicate questions concerning chemical identity, chemical concentration, or both. Because the data validation process is intended to assess the effect of QC issues on data usability, validation data qualifiers are attached to the data subsequent to the laboratory qualifiers and supersede any laboratory qualifiers. During the 1998 field investigation, QA/QC sample results consisting of equipment rinsate blanks were used to qualify the appropriate environmental sample results while field blank results were used to qualify all environmental sample results.

Qualified data are flagged with a letter qualifier representing an associated explanatory note needed to clarify the corresponding analytical result. Data qualified as "J" were retained as estimated. Estimated analytical results within a data set are common and considered to be usable by the USEPA (USEPA, 1989b). Data may be qualified as estimated for several reasons including an exceedence of holding times, high or low surrogate recovery, or intra-sample variability. In addition, values may be assigned an estimated "J" qualifier if the reported value is below the Contract Required Detection Limit (CRDL) or the Contract Required Quantitation Limit (CRQL). Data assigned a rejected, "R" qualifier, were excluded from the usable data set, and other qualified data were retained in the usable data set. Under these conditions, estimated positive results were designated with "J" qualifiers and rejected data were assigned "R" qualifiers. A qualifier of "B" was used if the analyte was detected in the samples associated blank. Data assigned a qualifier of "K" indicates that the reported value may be biased high, and a qualifier of "L" indicates the reported value may be biased low. Data qualifier definitions are summarized in Appendix F. Annotated Form I's for all samples were reviewed by the data validator and included in the overall data assessment. Based on the overall assessment of this data, the validator has deemed it accurate and representative of site conditions.

## **5.2 Data Management and Tracking**

The management and tracking of environmental samples from the time of field collection to the receipt of the validated electronic analytical results is of primary importance and reflects the overall quality of the laboratory results. Field samples and their corresponding analytical tests were recorded on the chain-of-custody sheets, which are included in Appendix C. The chain-of-custody records were compared to the sampling plan to determine if all designated samples were collected for the appropriate parameters. Similarly, the validated information was compared to laboratory information as a final check. In summary, the tracking information was used to identify the following items:

- Identify sample discrepancies between the sampling plan and the field investigation.
- Verify that the laboratory received all samples, and analyzed for the correct parameters.
- Verify that the data validator received a complete data set.
- Ensure that a complete data set was available for each media of concern prior to entering results into the database.

## **5.3 Non-Site Related Analytical Results**

Some organic and inorganic constituents detected in soil, groundwater, surface water, and sediments at Site 1 can be attributed to non-site related conditions or activities. Two primary sources of non-site related results include laboratory contaminants and naturally occurring inorganic elements. In addition, non-site related operational activities and conditions might contribute to "on-site" contamination. A discussion of non-site related analytical results is provided in the subsections that follow, and includes laboratory contaminants, non-site related contaminants, and naturally occurring inorganic elements.

Blank samples provide a measurement of contamination that has been introduced into a sample set during the collection, transportation, preparation, and/or analysis of samples. To remove non-site related contaminants from further consideration, the concentrations of chemicals detected in blanks were compared with concentrations of the same chemicals detected in environmental samples. QA/QC sample data for the RI investigations is included herein.

Additionally, available QA/QC data from previous, non-RI investigations is also included. QA/QC data associated with the 1992 soil samples, 1998 soil, surface water, and sediment samples, 1999 soil samples is included. Specifically, these samples include, trip blanks, rinsate blanks, and field blanks. Field blanks were collected to assess ambient conditions at the site during a given sampling event.

Common laboratory contaminants (i.e., acetone, 2-butanone, methylene chloride, toluene, and phthalate esters) were considered as positive results only when observed concentrations exceeded ten times the maximum concentration detected in any blank. If the concentration of a common laboratory contaminant was less than ten times the maximum blank concentration, then it was concluded that the chemical was not detected in that particular sample (USEPA, 1989). The maximum concentrations of detected common laboratory contaminants for each sampling event in blanks for the site were as follows:

1992

Acetone 24 µg/L (sample 1SB07-1A)

1998

Bis(2ethylhexyl)phthalate 1J µg/L (sample 388-RS03)

1999

Acetone 8J µg/L (sample 104-RS04)

Methylene Chloride 8J µg/L (sample 104-TB04)

2000

Acetone 3J µg/L (sample 00-POND-RS04)

Methylene Chloride 1J µg/L (sample 00-POND-RS04)

## 2001

Methylene chloride                      2J µg/L (sample CX01-TB03)  
Bis(2-ethylhexyl)phthalate            2J µg/L (sample 1116RB)

QA/QC sample analytical results are included in Appendix G.

Other constituents contained in blanks that are not considered common laboratory contaminants were considered as positive results only when observed concentrations exceeded five times the maximum concentration detected in any blank (USEPA, 1989). All TCL compounds of less than five times the maximum level of contamination noted in any blank were considered to be not detected in that sample. The maximum concentrations of all other detected blank contaminants were as follows:

## 1992

Chloroform                                      5J µg/L (sample 1SB07-1C)

## 1998

Chloroform                                      9Jµg/L (sample 388-RS03)

QA/QC sample analytical results are included in Appendix G. Several metals were detected in field and equipment rinsate blanks from 1992, 1998, 1999, and 2000. Concentrations were generally orders-of-magnitude lower in the blank samples than in environmental samples. Thus, the presence of metals in blank samples has no impact on the interpretation of the data. QA/QC sample analytical results are included in Appendix G.

## **5.4    State and Federal Criteria and Standards**

Contaminant concentrations can be compared to contaminant-specific established State and Federal criteria and standards such as Maximum Contaminant Levels (MCLs).

The only enforceable Federal regulatory standards for water are the Federal MCLs. Regulatory guidelines were used for comparative purposes to infer the potential risks and environmental

impacts when necessary. Relevant regulatory guidelines include Region III Risk Based Concentrations (RBCs) and direction for USEPA Toxicology. A brief explanation of the criteria and standards used for the comparison of site analytical results is presented below:

**Region III Risk Based Concentrations (RBCs)** - April 2000 - RBC values are derived using conservative USEPA promulgated default values and the most recent toxicological criteria available. The RBCs for potentially carcinogenic chemicals are based on a target Incremental Cancer Risk (ICR) of  $1 \times 10^{-6}$ . The RBCs for non-carcinogens are based on a target hazard quotient of 1.0. For potential carcinogens, the toxicity criteria applicable to the derivation of RBC values are oral and inhalation cancer slope factors (CSFs); for non-carcinogens, they are chronic oral and inhalation reference doses (RfDs). These toxicity criteria are subject to change as more updated information and results from the most recent toxicological/epidemiological studies become available. Therefore, the use of toxicity criteria in the derivation of RBC values requires that the screening concentrations be updated periodically to reflect changes in the toxicity criteria. The RBC table is issued on a semi-annual basis.

**Federal Maximum Contaminant Levels** - July 2002 - Federal MCLs are enforceable standards for public water supplies promulgated under the Safe Drinking Water Act (SDWA) and are designed for the protection of human health. MCLs are based on laboratory or epidemiological studies and apply to drinking water supplies consumed by a minimum of 25 persons. They are designed for prevention of human health effects associated with lifetime exposure (70-year lifetime) of an average adult (70-kg) consuming 2 liters of water per day. MCLs also consider the technical feasibility of removing the contaminant from the public water supply.

**Surface Water Screening Criteria** - Surface water screening criteria were based on the recommendation of USEPA Toxicology on August 26, 1998. It was recommended that the maximum detection be compared to ten times the Tap Water RBC.

**Sediment Screening Criteria** - Sediment screening criteria were based on the recommendation of USEPA Toxicology on August 26, 1998. It was recommended that the maximum detection be compared to ten times the Residential RBC.

## **5.5 Nature and Extent of Contamination**

This section provides the analytical data for all media sampled. The discussion is organized by media as follows: surface soil, subsurface soil, groundwater, surface water, and sediment. Table 4-1 provides a summary of all the samples used to determine the nature and extent of contamination. As mentioned in Section 4.0, samples from non-RI investigations were included in the data set where appropriate.

### **5.5.1 Surface Soil**

Five surface soil samples (and two duplicate samples) were collected within the landfill area between 1992 and 1999 and have been included in this RI Report. Three additional soil samples were collected within the landfill area in 2001 for ecological risk screening purposes and will be included in this discussion. One surface soil sample was collected outside the landfill area in 1992 that has also been included in this RI Report. An additional seven soil samples were collected in November of 2001 outside of the landfill area for ecological risk screening purposes. Data summaries are presented on Tables 5-1 through 5-4.

Table 5-1 presents a summary of organic compound detections from samples within the landfill area. Detections of VOCs were limited to low concentrations of methylene chloride and methyl acetate in one sample. PAHs, pesticides, and two PCB congeners (Aroclor-1260 and Aroclor-1248) were detected in most of the soil samples. Data from sample 1-HA01-00 exhibited the highest detections, eight compounds were over screening criteria as shown on Figure 5-1.

Figure 5-1 shows the distribution of organics in surface soil above screening criteria. Based on the soil samples collected, it appears that the highest concentrations of PAHs, pesticides, and PCBs occur along the western side of the landfill. It was reported in Baker's 1998 Field Investigation Report (Baker 1999) that these surface soil samples were collected from within the landfill ash. As this landfill is known to contain a variety of wastes including ash and miscellaneous solid waste, it is apparent that the results of surface soil samples collected in the landfill ash area represent the contents of the landfill rather than cover soil. Further, the presence of SVOCs in these samples (which are primarily PAHs) is expected. PAHs are products of combustion and incomplete combustion of carbonaceous materials (often resulting in ash) and can be found in certain petroleum products, creosote, road tar, mineral oils, coal tar, and soot. Certain PAHs have also been associated with wood preserving, paints, inks, pharmaceuticals, and

photographic materials. Therefore, the presence of PAHs in samples collected from within the limits of the landfill is expected.

Based on the known and potential unknown contents of the landfill, it also appears as if some of the material disposed in the landfill contained pesticides at low concentrations. This could be the result of yard waste and/or clippings that might have been disposed in this landfill. Similarly, the presence of Aroclor-1260 and Aroclor-1248 in a few of these samples may be the result of materials disposed in this landfill.

Table 5-2 presents a summary of inorganic compound detections in surface soil samples collected within the landfill area. Figure 5-2 shows the distribution of inorganics in surface soil above screening criteria. Antimony, arsenic, copper, iron, and lead exceed screening criteria. These compounds might have been present in materials (including ash) that were disposed in the landfill. It would be expected that the presence of lead, copper, and iron would be associated with landfilled materials. All are common in a wide variety of products. For example, lead is often used in some types of batteries, ammunition, paints, ceramics, ballast or other weights, solder, pipes, gasoline additives, electrical parts of various types of machinery, and circuit boards. Copper is often used in plumbing materials and electrical parts, and iron would be associated with any steel product. Antimony and arsenic are less common, but have anthropogenic associations as well (Hem, 1992). Based on the soil samples collected, it appears that the highest concentrations of inorganics occur along the western side of the landfill.

Table 5-3 presents a summary of organic compound detections in surface soil samples collected outside the landfill area. Detections of VOCs were limited to methylene chloride, methyl acetate and 1,1,2,2-tetrachloroethane. SVOCs were detected in low concentrations; however, none of the detected compounds exceeded screening criteria as shown in Table 5-3. There were no exceedances of Pesticides/PCBs above screening criteria.

Table 5-4 presents a summary of inorganic compound detections in surface soil samples collected outside the landfill area. All of the inorganic compounds analyzed for were detected in the sample outside the landfill area. Detections were generally less in the sample outside the landfill area than inside the landfill area, including antimony, arsenic, copper, iron, and lead. Of the inorganic compounds detected, arsenic exceeded screening criteria in all samples, as shown on Figure 5-2. Iron was detected in concentrations above screening criteria in two samples as shown on Figure 5-2.

### 5.5.2 Subsurface Soil

Nine subsurface soil samples (and two duplicate samples) were collected within the landfill area between 1992 and 1999 that have been included in this RI. Of the nine samples, two were collected from native soil below the landfilled material (1-DPB07-03 and 1-DPB09-03). Three samples were collected from landfilled ash material, including 1-DPB02-02, 1-DPB06-02, and 1-DPB10-02. Additionally, two subsurface soil samples were collected outside the landfill area in 1992 and also have been included in this RI (1-SB08-2A and 1-SB08-3A). An additional four samples (and a duplicate sample) were collected outside of the landfill area at depths between 0.5 and 2.0 feet in August 2001, for ecological risk screening purposes.

Table 5-5 presents a summary of organic compound detections in subsurface soil samples collected within the landfill area. Figure 5-3 shows the distribution of organics in subsurface soil above screening criteria. Five VOCs were detected in subsurface soil samples at low levels and were below screening criteria. PAHs, pesticides, and PCBs were detected in most of the samples. Ash samples 1-DPB02-02 and 1-DPB06-02 exhibit the highest concentrations of PAHs. It should be noted that the PAHs detected in subsurface samples were generally the same as detected in surface samples. Some pesticides were also detected in the ash samples. Explosive compounds were detected only in subsurface ash samples at levels above screening criteria. Aroclor-1260 and 4,4-DDT were detected at elevated concentrations in sample 1-HA02-02. This sample is located just south of the large debris pile. Native soil sample 1-DPB07-03 collected beneath landfilled materials did not exhibit any detections of any organic compounds. Native soil sample 1-DPB09-03 collected beneath landfilled materials exhibited only low detections of a few PAH compounds, benzo(a)pyrene was detected above USEPA residential RBCs.

Table 5-6 presents a summary of inorganic compound detections in subsurface soil samples collected within the landfill area. Figure 5-4 shows the distribution of inorganics in subsurface soil above screening criteria. Antimony, arsenic, copper, iron, and lead exceed screening criteria. This pattern of exceedences is consistent with surface soil samples, and is expected with landfilled materials. Some metals might not volatilize during incineration and might accumulate in residual material (i.e., ash). It is apparent that the landfilled materials have not significantly impacted native soils below. A qualitative comparison suggests that native soil samples below landfilled materials exhibit inorganic concentrations similar to that of other soils outside the landfill area.

Table 5-7 presents a summary of organic compound detections in the six subsurface soil samples collected outside the landfill area. Detections of VOCs were limited to low levels of acetone, methyl acetate and methylene chloride. It should be noted that acetone was detected in the trip blank at levels comparable to the environmental sample. Organic compounds were detected in low concentrations in samples outside the landfill area, as shown on Table 5-7. Benzo(a)pyrene was the only SVOC detected that exceeded screening criteria in sample CX01-SB02.

Table 5-8 presents a summary of inorganic compound detections in subsurface soil samples collected outside the landfill area. All inorganic compounds analyzed were detected. With the exception of potassium, concentrations were lower in subsurface soil samples outside the landfill area than samples within the landfill area. Arsenic however, still exceeded screening criteria.

### **5.5.3 Groundwater**

Groundwater samples were collected from five monitoring wells in May 2000. Table 5-9 shows a summary of organic compound detections in groundwater, and Table 5-10 shows a summary of total and dissolved inorganic compound detections.

No VOCs were detected in the groundwater samples. Table 5-9 shows that three phthalate compounds and two explosive compounds were detected at trace levels. Only bis(2-ethylhexyl)phthalate, 4-Nitrotolulene, Di-n-octylphthalate and Di-n-octylphthalate exceeded screening criteria (Figure 5-5). Two of the three phthalate compounds (bis(2-ethylhexyl)phthalate, di-n-octylphthalate), and one explosive compound (HMX) were detected in the upgradient sample, 1-GW05-00. Additionally, phthalate compounds are common sampling and laboratory artifacts. This is evidenced by the fact that bis(2-ethylhexyl)phthalate was detected in the field and equipment rinsate blanks as well as the associated laboratory internal blanks.

Table 5-10 shows detections of total and dissolved inorganic compounds. Figure 5-6 shows the distribution of inorganics in groundwater above screening criteria. Only total aluminum, arsenic, iron, and manganese exceeded screening criteria. Total iron and manganese were detected in the upgradient sample as well. Except for well GW07, total iron was also higher in the upgradient sample. Detections of arsenic in the four downgradient wells and iron in well GW07 may be related to the landfill. Arsenic and iron in well GW07 are one to two orders-of-magnitude higher

than all other wells. It should be noted that this well is screened immediately below the landfill. Dissolved arsenic, iron and manganese were detected above tap water RBCs and background values as shown in Table 5-10.

#### **5.5.4 Surface Water**

Three surface water samples were collected as part of the 1998 Field Investigation. An additional four samples (and one duplicate sample) were collected in August 2001 for ecological risk screening. Table 5-11 shows a summary of organic compound detections in surface water, and Table 5-12 shows a summary of inorganic compound detections.

Table 5-11 shows that bis(2-ethylhexyl)phthalate was detected at low levels in all three samples, including the upstream sample location. Bis(2-ethylhexyl)phthalate was detected in a QA/QC sample at the same concentration and is therefore not considered to be site related. Methyl acetate was detected below screening criteria in the duplicate sample for CX01-SW04. No figure was generated for organic compounds because there were no exceedences of screening criteria.

Table 5-12 shows that most inorganic compounds were detected in surface water samples. Antimony, arsenic, copper, iron, and lead were identified in the landfill material. Of those inorganics, only arsenic exceeds screening criteria in surface water and is shown on Figure 5-7. From an examination of Table 5-13, the trends of these four remaining compounds show that maximum detections occur in the sample located adjacent to the landfill, namely 1-SW02.

#### **5.5.5 Sediment**

Four sediment samples were collected as part of the 1998 Field Investigation. An additional 24 sediment samples (including 2 duplicate samples) were collected in August 2001 for ecological risk screening purposes. Both the 1998 and the 2001 data have been incorporated in the RI report. Table 5-13 shows a summary of organic compound detections in sediment, and Table 5-14 shows a summary of inorganic compound detections.

Table 5-13 shows the detection of only two VOCs, but several detections of PAHs, pesticides, and one PCB congener (Aroclor-1260). Figure 5-8 shows organic compound concentrations in sediment above screening criteria. Only benzo(a)pyrene exceeds screening criteria, in two samples (1-SD02 and 1-SD04).

Table 5-14 shows that antimony, arsenic, copper, iron, and lead were detected in sediment samples. Except for arsenic all detections in sediment samples were below screening criteria. Arsenic exceeds screening criteria at seven sampling locations and is shown on Figure 5-9. Arsenic was detected at higher concentrations from the samples collected in the debris area.

## **5.6 Hazardous Waste Characteristics of Landfilled Materials**

Appendix H shows the analytical results for hazardous waste characteristics of select samples of landfilled materials. The analytical results suggest that the composite samples of landfilled ash are not hazardous by characteristic. Two inorganic compounds were detected in the TCLP leachate of the ash, but below the Toxicity Characteristic. Barium was detected at 1,270 µg/L in sample 1-DPB04-COMP, and lead at 922 µg/L in sample 1-DPBCOMP-01. The Toxicity Characteristic for barium is 100,000 µg/L, and is 5,000 µg/L for lead Characteristic (40 Code of Federal Regulations [CFR] 261.24). No organic compounds were detected in the TCLP leachate. Additionally, none of the samples are hazardous by characteristic for reactivity, corrosivity, or ignitability (40 CFR 261.21-23).

One of the three miscellaneous debris samples exceeded the Toxicity Characteristic for lead. Lead was detected at 6,480 µg/L in sample 01-TCRA03. No organic compounds were detected in the TCLP leachate. Additionally, none of the samples are hazardous by characteristic for reactivity, corrosivity, or ignitability. The results suggest that some of the landfilled materials are hazardous by characteristic.

## **5.7 Geotechnical Samples**

The results from the two soil samples collected for geotechnical analysis are included in Appendix I. The results of the direct shear analysis will be used for future remedial design that might include bank improvements.

## **5.8 Summary**

The landfill contains a variety of wastes that are both non-hazardous and wastes that exhibit hazardous characteristics. The analytical data presented show that samples of landfilled material exhibit the presence of PAHs, pesticides, PCBs, and inorganic compounds, particularly antimony,

arsenic, copper, iron, and lead. The data shows evidence suggesting that the landfill has impacted the surrounding environment to a limited extent.

The data suggest that organic and inorganic constituents from landfilled material have not significantly impacted native soils surrounding and beneath the landfill. Low levels of PAHs in native soil samples support this conclusion. Fewer PAHs were detected in native soils and at concentrations well below screening criteria and below concentrations in samples from landfilled material. Inorganic compounds such as antimony, arsenic, copper, iron, and lead are naturally-occurring compounds. These compounds were detected in native soil samples at concentrations below concentrations in samples from landfilled material, and except for arsenic, below screening criteria. It should be noted that because arsenic is relatively toxic, screening criteria such as RBCs are very low. Based on background soil sampling programs at other LANTDIV Continental United States and Caribbean facilities, background levels of arsenic often exceed RBCs.

There is limited evidence that groundwater has been impacted by the landfill. Organic compounds (phthalates and explosives) were detected in groundwater samples. Arsenic and iron were detected in downgradient groundwater samples above background and screening criteria. Both arsenic and iron are present in samples of landfilled materials.

There is evidence that the landfill has impacted sediment to a limited extent. Sediment data show an increase concentration from upstream to downstream locations. Additionally, sediment samples collected from locations adjacent to the landfill exhibit the some of the highest detections of organics and inorganics, as well as the highest number of maximum detections. However, it is important to note that only a few compounds exceeded screening criteria.

Non-naturally occurring contaminants identified at Site 1 (PAHs, pesticides, and PCBs) are chemically immobile. These compounds can exhibit limited mobility due to erosion of soil particles on which these compounds adsorbed. The analytical data support this; PAHs, pesticides, and PCBs were not detected in surface water or groundwater, but have accumulated in sediments at low concentrations.

## 6.0 CONTAMINANT FATE AND TRANSPORT

The potential for a contaminant to migrate and persist in an environmental medium is critical when evaluating the potential for a chemical to elicit an adverse human health or ecological effect. The environmental mobility of a chemical is influenced by its physical and chemical properties, the physical characteristics of the site, and the site chemistry. This section presents a discussion of the various physical and chemical properties of contaminants detected at Site 1, and their fate and transport through the environment.

### 6.1 Chemical and Physical Properties

Table 6-1 presents the physical and chemical properties associated select organic contaminants detected during the investigations. These properties determine the inherent environmental mobility and fate of a contaminant. These properties include:

- Vapor Pressure
- Water Solubility
- Octanol/Water Partition Coefficient
- Organic Carbon Adsorption Coefficient (sediment partition)
- Specific Gravity
- Henry's Law Constant
- Mobility Index

A discussion of the environmental significance of each of these properties is discussed in the subsections that follow.

Vapor Pressure provides an indication of the rate at which a chemical may volatilize. It is of primary significance at environmental interfaces such as surface soil/air and surface water/air. Volatilization can be important when evaluating groundwater and subsurface soils, particularly when selecting remedial technologies. Vapor pressure for monocyclic aromatics are generally higher than vapor pressures for polyaromatic hydrocarbons (PAHs). Contaminants with higher vapor pressures (e.g., VOCs) will enter the atmosphere at a quicker rate than the contaminants with low vapor pressures (e.g., inorganics).

Water Solubility is proportional to the rate at which a contaminant is leached from the soil by infiltrating precipitation. More soluble contaminants (e.g., VOCs) are usually more readily leached than less soluble contaminants (e.g., inorganics). The water solubilities indicate that the volatile organic contaminants including monocyclic aromatics are usually several orders-of-magnitude more soluble than PAHs. Consequently, highly soluble compounds such as the chlorinated VOCs will migrate at a faster rate than less water-soluble compounds.

Octanol/Water Partition Coefficient ( $K_{ow}$ ) is the ratio of the chemical concentration in octanol divided by the concentration in water. The octanol/water partition coefficient has been shown to correlate well with bioconcentration factors in aquatic organisms and adsorption to soil or sediment. Specifically, a linear relationship between octanol/water partition coefficient and the uptake of chemicals by fatty tissues of animal and human receptors (the bio-concentration factor [BCF]) has been established (Lyman, 1982). The coefficient is also useful in characterizing the sorption of compounds by organic soils where experimental values are not available.

The Organic Carbon Adsorption Coefficient ( $K_{oc}$ ) indicates the tendency of a chemical to adhere to soil particles organic carbon. The solubility of a chemical in water is inversely proportional to the  $K_{oc}$ . Contaminants with high soil/sediment adsorption coefficients generally have low water solubilities. For example, contaminants such as PAHs are relatively immobile in the environment and are preferentially bound to the soil. These compounds are not subject to aqueous transport to the extent of compounds with higher water solubilities. Erosional properties of surface soils may, however, enhance the mobility of these bound soil contaminants.

Specific Gravity is the ratio of a given volume of pure chemical at a specified temperature to the weight of the same volume of water at a given temperature. Its primary use is to determine whether a contaminant will have a tendency to "float" or "sink" (as an immiscible liquid) in water.

Henry's Law Constant is a relationship between pressure and solubility. Vapor pressure and water solubility are of use in determining volatilization rates from surface water bodies and from groundwater. These two parameters can be used to estimate an equilibrium concentration of a contaminant in the water phase and in the air directly above the water, this can be expressed as Henry's Law.

Mobility Index is a qualitative assessment that takes into consideration water solubility (S), vapor pressure (VP), and organic carbon partition coefficient ( $K_{oc}$ ). It is defined as:

$$MI = \log[(S \cdot VP) / K_{oc}]$$

A scale to evaluate MI has been prepared by Ford and Gurba (Ford, 1984) and is presented below:

<u>Relative MI</u>	<u>Mobility Description</u>
>5	Extremely Mobile
0 to 5	Very Mobile
-5 to 0	Slightly Mobile
-10 to -5	Immobile
<-10	Very Immobile

## 6.2 Contaminant Transport Pathways

Based on the evaluation of existing conditions at Site 1, the following potential contaminant pathways have been defined.

- Surface soil runoff.
- Leaching of soil contaminants to groundwater.
- Migration of groundwater contaminants.
- Leaching of sediment contaminants to surface water.
- Migration of contaminants in surface water.

Contaminants released to the environment could also undergo the following during transportation:

- Physical transformations: volatilization, precipitation
- Chemical transformations: photolysis, hydrolysis, oxidation, reduction
- Biological transformations: biodegradation
- Accumulation in one or more media

The paragraphs that follow describe the potential transport pathways listed above.

### **6.3 Surface Soil Runoff**

Water can erode exposed soil particles during precipitation events. This is influenced by site topography, the amount of precipitation, soil particle size/density and cohesion, and vegetative cover. There is a relatively steep slope between the landfill and the unnamed tributary to the York River and York River itself. This slope has eroded in the past leaving landfill contents exposed. A TCRA along the York River slope was conducted to address this problem. Surface soil runoff is a significant pathway.

### **6.4 Leaching of Soil Contaminants to Groundwater**

The contaminants present in the soil samples at Site1 are primarily PAHs, pesticides, and PCBs. Contaminants that adhere to soil particles or have accumulated in pore spaces can leach and migrate vertically to the groundwater as a result of the infiltration of precipitation. The rate and extent of leaching is influenced by the depth to the water table, amount of precipitation, rate of infiltration, the physical and chemical properties of the soil and the contaminants.

Groundwater samples were collected from shallow monitoring wells. The groundwater analytical results can be compared to soil sample analytical results to determine if contaminants detected in soil have migrated or may migrate in the future, to underlying groundwater. These results were discussed in detail in Section 5.0.

PAHs, pesticides, and PCBs were not in groundwater samples. The mobility index on Table 6-1 shows that these compounds are "immobile" to "very immobile". Thus, it is expected that these compounds would not readily leach from soil to groundwater.

The mobility of inorganic compounds is controlled by different factors. Table 6-2 presents relative mobilities of inorganic compounds under varying environmental conditions. Under oxidizing and acidic conditions, which are common in shallow East Coast aquifers, inorganic compounds such as iron and lead are not very mobile.

## **6.5 Migration of Groundwater Contaminants Offsite**

Contaminants leaching from soils to underlying groundwater can migrate as dissolved constituents in groundwater in the direction of groundwater flow. Three general processes govern the migration of dissolved contaminants caused by the flow of water: (1) advection, movement caused by flow of groundwater; (2) dispersion, movement caused by irregular mixing of waters during advection; and (3) retardation, principally chemical mechanisms which occur during advection.

The potential for contaminants leaching from soil to groundwater is low, as discussed in Section 6.4. The analytical data tend to confirm that, as site-related contaminants have not been observed in groundwater to any significant extent. Thus, it appears that potential for migration of contaminants offsite in groundwater is low.

## **6.6 Leaching of Sediment Contaminants to Surface Water**

Contaminants that adhere to sediment particles or have accumulated in sediment pore spaces can leach and migrate to the surface water. The physical and chemical properties of the soil and the physical and chemical properties of the contaminant influence the rate and extent of this migration.

## **6.7 Migration of Contaminants in Surface Water**

Contaminants leaching from soils to surface water can migrate as dissolved constituents in surface water in the direction of surface water flow. Three general processes govern the migration of dissolved contaminants caused by the flow of water: (1) movement caused by the flow of surface water, (2) movement caused by the irregular mixing of water, and (3) chemical mechanisms occurring during the movement of surface water. Sediment particles can also disassociate from the sediment into surface water and migrate by one of the aforementioned methods.

## 7.0 HUMAN HEALTH RISK ASSESSMENT

A baseline human health risk assessment (HHRA) was performed to evaluate the potential risks associated with exposure to environmental media at Site 1 - Landfill Near Incinerator at Naval Weapons Station Yorktown (WPNSTA), Yorktown, Virginia, Cheatham Annex Site (CAX) if remedial action is not undertaken. The baseline HHRA considers the most likely routes of potential human exposure for both current and future risk scenarios. The baseline HHRA was conducted in accordance with the Risk Assessment Guidance for Superfund (RAGS), Part A, Human Health Evaluation Manual (United States Environmental Protection Agency [USEPA], 1989), and the most recent updates, including the reporting format as set forth in RAGS Part D (USEPA, 1998b). The complete set of RAGS Part D tables is presented in Appendix L. A discussion of the previous investigations and history of Site 1 is included in Section 2.0.

Section 7.0 describes the HHRA based on evaluation of the data collected from previous investigations (i.e., 1992 Site Investigation and 1998 Field Investigation), data collected for the Remedial Investigation (RI) during the November 1999 Field Investigation, May 2000 groundwater sampling effort, and the November 2001 sampling event. Specifically, data evaluated for Site 1 includes surface soil (1998 and 1999 Field Investigations and 2001 sampling event), subsurface soil (1992 Site Investigation, 1998 and 1999 Field Investigations, and 2001 sampling event), groundwater (2000 groundwater sampling effort), surface water (1998 Field Investigation and 2001 sampling event), and sediment (1998 Field Investigation and 2001 sampling event).

The baseline HHRA is comprised of seven sections; Section 7.1 presents the selection of Chemicals of Potential Concern (COPCs). Sections 7.2 and 7.3 present the Exposure Assessment and Toxicity Assessment, respectively. The Risk Characterization is presented in Section 7.4. Section 7.5 presents sources of uncertainty inherent in the estimation of inferential potential human health effects. Summary and conclusion of the baseline HHRA is provided in Section 7.6. Section 9.0 presents the references.

### 7.1 Identification of Chemicals of Potential Concern

The selection of COPCs was based on the information provided in the USEPA Region III Technical Guidance on Selecting Exposure Routes and Contaminants of Concern, by Risk-Based

Screening, dated January 1993 (USEPA, 1993c) and USEPA's Risk Assessment Guidance for Superfund, Volume I. Human Health Evaluation Manual (Part A), Interim Final, December 1989 (USEPA, 1989). COPC selection was completed for each environmental medium using analytical data obtained during this RI and previous investigations.

A discussion of laboratory analytical results and nature and extent of constituent contamination was presented in Section 5.0 of this report. Chemicals detected in environmental media sampled during the RI were re-evaluated in this section to select COPCs for quantitative evaluation in the baseline HHRA. Chemicals selected as COPCs that could not be quantitatively evaluated are discussed in the uncertainties section (Section 7.5) of the baseline HHRA.

#### **7.1.1 COPC Selection Criteria**

The primary criterion used in selecting chemicals as a COPCs at Site 1 included comparing maximum detected sample concentrations to the USEPA Region III Risk-Based Concentration (RBC) Values (USEPA, 2002), in accordance with USEPA Region III Selection of Contaminants of Concern by Risk-Based Screening (SCCRBS) guidance (USEPA, 1993c).

In conjunction with concentration comparisons to the USEPA Region III RBC values, a comparison to concentrations detected in field and laboratory blanks was conducted by a third-party validator, to ensure that only site-related contaminants are evaluated in the quantitative estimation of human health effects. Furthermore, those constituents generally considered to be essential nutrients (calcium, magnesium, potassium, and sodium, which have relatively low toxicity) were not evaluated in this baseline HHRA.

The toxicity of a chemical detected in a given environmental medium, as well as the history of site-related activities are other important criteria applied in selecting COPCs at Site 1. Therefore, in conjunction with concentration comparisons to USEPA Region III RBC values, evaluations of toxicity and site history were considered to determine whether chemicals eliminated by a direct comparison to RBC values should be re-included as COPCs. Each of the aforementioned criteria is discussed in the paragraphs that follow.

**USEPA Region III RBC Values** - Region III RBC values were derived using conservative USEPA promulgated default values and the most recent toxicological criteria available. RBCs for potentially carcinogenic and noncarcinogenic chemicals were individually derived based on a

target Incremental Lifetime Cancer Risk (ILCR) of  $1 \times 10^{-6}$  and a target Hazard Quotient (HQ) of 1.0, respectively. For potential carcinogens, the toxicity criteria applicable to the derivation of the RBC are oral and inhalation cancer slope factors; for noncarcinogens, they are chronic oral and inhalation reference doses. For noncarcinogens, each RBC value was reduced by a factor of 10 to ensure that chemicals with additive effects are not prematurely eliminated during screening (USEPA, 1993c).

In this baseline HHRA, constituents in groundwater will be compared to tap water RBC values, constituents identified in soil and sediment will be compared to residential soil RBCs, and constituents detected in surface water will be compared to tap water RBCs times ten (recommended by Region III toxicologist). (Baker, 2001).

**Blank Concentrations** - If a chemical is detected in both the environmental sample and a blank sample, it may not be retained as a COPC in accordance with RAGS depending on the concentration of the chemical in the media. Therefore, blank data were compared with results from environmental samples. If the blanks contained detectable results for common laboratory contaminants (i.e., acetone, 2-butanone, methylene chloride, toluene, and phthalate esters), environmental sample results were considered as positive results only if they exceed 10 times the maximum amount detected in the associated blank. If the chemical detected in the blank(s) is not a common laboratory contaminant, environmental sample results were considered as positive results only if they exceeded five times the maximum amount detected in the associated blank(s). Furthermore, the elimination of an environmental sample result would directly correlate to a reduction in the prevalence of the contaminant in that media.

Associated blanks collected during the field investigation for Site 1 included field (water source) blanks, trip blanks, and rinsate blanks. The aforementioned methodologies for evaluating blanks were implemented during independent third party analytical data validation prior to the selection of COPCs in the risk assessment.

**Essential Nutrients** - Despite their inherent toxicity, certain inorganic constituents are essential nutrients. Essential nutrients need not be considered further in the baseline HHRA if they are present in relatively low concentrations (i.e., slightly elevated above naturally occurring levels), or if the constituent is toxic at doses much higher than those which could be assimilated through exposures at the site (USEPA, 1989). Elements evaluated as essential nutrients include calcium,

magnesium, potassium and sodium. Although iron is considered an essential nutrient, it is evaluated quantitatively in this HHRA since toxicity criteria are available for this analyte.

### **7.1.2 Re-inclusion of Chemicals as COPCs**

Chemicals can be re-included as COPCs for quantitative evaluation in the baseline HHRA, despite having been eliminated as such from a comparison to RBC values (or other aforementioned criteria). For example, a chemical that was detected at concentrations below the corresponding RBC value may be re-included as a COPC if a chemical is considered a Class A carcinogen (human carcinogen), or if it is reasonable to assume that the chemical could be site-related (especially if it has been detected in other media of concern). In addition, all carcinogenic Polyaromatic Hydrocarbons (cPAHs) detected in a medium of concern are re-included as COPCs if only one cPAH is retained as a COPC in that medium based on the potential additive toxic effects of cPAHs.

### **7.1.3 Selection of COPCs**

Four environmental media (soil, groundwater, surface water, and sediment) were investigated at Site 1. The data used in this HHRA for Site 1 was comprised of data collected for the 1992 Site Investigation, 1998 Field Investigation, 1999 Field Investigation, the 2000 groundwater sampling event, and the 2001 sampling event.

The selection of soil COPCs at Site 1 was stratified to include the surface soil (0- to 6-inches and 0- to 2- feet Below Ground Surface [bgs]) and the subsurface soil (depths greater than 1 foot bgs) by depth interval. The 0- to 2- foot interval was included in the surface soil data set because these samples were collected to characterize the existing soil cap (i.e., "surface" material). Groundwater COPCs were selected from samples collected from the Yorktown Eastover aquifer. Site 1 surface water and sediment data were obtained from samples collected from the unnamed tributary to the York River. This baseline HHRA selects COPCs from the following analytical parameters: volatile organic compounds (VOCs), semivolatile organic compounds (SVOCs), pesticides, PCBs, nitramine compounds, and inorganics (metals). Furthermore, calcium, magnesium, potassium, and sodium were detected in almost every sample, regardless of the medium; however, these constituents were considered to be essential nutrients and were therefore, not retained as COPCs in any medium under investigation at Site 1.

Tables 7-1 through 7-7 present the selection of COPCs for each environmental medium based on comparison of the maximum detected concentration with the USEPA Region III RBC values and consideration of other applicable criteria (i.e., site history and chemical toxicity). The shaded areas in the tables indicate constituents retained as COPCs. Information is presented in these tables only for those constituents detected at least once in the medium of interest. Other statistical information is presented in Appendix J.

The following paragraphs present the rationale for selection of COPCs. Sample locations, analytical results, and corresponding figures are presented in other sections of this RI report.

#### 7.1.3.1 Surface Soil Inside Landfill

During the 2001 sampling event, three surface soil samples were collected inside the landfill and analyzed for VOCs, SVOCs, nitramine compounds, pesticides, PCBs, and inorganics. During the 1999 Field Investigation, two surface soil samples were collected inside the landfill and analyzed for VOCs, SVOCs, nitramine compounds, pesticides, PCBs, and inorganics. During the 1998 Field Investigation at Site 1, three surface soil samples (2 environmental and 1 duplicate) were collected inside the landfill from the 0- to 6-inch interval and analyzed for VOCs, SVOCs, pesticides, PCBs, nitramine compounds, and inorganics. During the 1992 Site Investigation, 2 surface soil samples from Site 1 were collected inside the landfill and analyzed for VOCs, SVOCs, and inorganics. Nitramine compounds were not detected in these surface soil samples. The COPC selection summary for the surface soil inside the landfill is presented on Table 7-1.

There were no VOCs detected in the surface soil inside the landfill at concentrations above corresponding residential soil RBC values. Therefore, VOCs were not retained as COPCs for surface soil inside the landfill.

Benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, carbazole, chrysene, dibenzo(a,h)anthracene, indeno(1,2,3-cd)pyrene, and phenanthrene were detected at concentrations above their respective residential soil RBC values and were retained as COPCs for surface soil inside the landfill.

4,4'-DDT and dieldrin were detected at concentrations above corresponding residential soil RBC values and were retained as COPCs for surface soil inside the landfill.

Aroclor-1248 and Aroclor-1260 were detected in the surface soil inside the landfill at Site 1 at concentrations exceeding corresponding residential soil RBC values and were retained as COPCs for surface soil inside the landfill.

Inorganics were detected in all surface soil samples collected inside the landfill at Site 1. Aluminum, antimony, arsenic, cadmium, chromium, copper, iron, lead, manganese, nickel, thallium, and zinc were detected at concentrations greater than residential soil RBC values and were retained as COPCs for surface soil inside the landfill. The results of the COPC selection for Site 1 surface soil inside the landfill are presented in Table 7-1.

#### 7.1.3.2 Surface Soil Outside Landfill

During the 2001 sampling event, nine surface soil samples were collected outside the landfill and analyzed for VOCs, SVOCs, nitramine compounds, pesticides, PCBs, and inorganics. During the 1992 Site Investigation at Site 1, one surface soil sample was collected outside the landfill and analyzed for VOCs, SVOCs, and inorganics. The COPC selection summary for the surface soil outside the landfill is presented on Table 7-2.

There were no VOCs detected in the surface soil outside the landfill at concentrations above corresponding residential soil RBC values. Therefore, VOCs were not retained as COPCs for surface soil outside the landfill.

Benzo(a)pyrene was detected at a maximum concentration above its respective residential soil RBC value and was retained as a COPC for surface soil outside the landfill. Benzo(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, carbazole, chrysene, dibenzo(a,h)anthracene, and indeno(1,2,3-cd)pyrene were detected at concentrations less than their respective residential soil RBC values but were retained as COPCs for surface soil outside the landfill because of the possible additive effect of cPAHs.

There were no pesticides detected in the surface soil outside the landfill at concentrations above their respective residential soil RBC values. Therefore, pesticides were not retained as COPCs for surface soil outside the landfill.

There were no PCBs detected in the surface soil outside the landfill at concentrations above residential soil RBC values. Therefore, PCBs were not retained as COPCs for surface soil outside the landfill.

Inorganics were detected in the surface soil sample collected outside the landfill at Site 1. Arsenic, chromium, copper, iron, manganese, and thallium were detected at concentrations greater than corresponding residential soil RBC values and were retained as COPCs for surface soil outside the landfill. The results of the COPC selection for the surface soil outside the landfill are presented on Table 7-2.

#### 7.1.3.3 Subsurface Soil Inside Landfill

During 1992 Site Investigation at Site 1, two subsurface soil samples were collected inside the landfill from depths greater than one foot bgs and analyzed for VOCs, SVOCs, and inorganics. During the 1998 Field Investigation, three subsurface soil samples (two environmental and one duplicate) from Site 1 were collected inside the landfill from depths greater than one foot bgs and analyzed for VOCs, SVOCs, nitramine compounds, pesticides, PCBs, and inorganics. During the 1999 Field Investigation five subsurface soil samples (four environmental and one duplicate) were collected inside the landfill from depths greater than one foot bgs and analyzed for VOCs, SVOCs, pesticides, PCBs, nitramine compounds, and inorganics. The COPC selection summary for the subsurface soil inside the landfill is presented on Table 7-3.

There were no VOCs detected in the subsurface soil outside the landfill at concentrations above corresponding residential soil RBC values. Therefore, VOCs were not retained as COPCs for subsurface soil inside the landfill.

Benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, dibenzo(a,h)anthracene, and indeno(1,2,3-cd)pyrene were detected at concentrations above their respective residential soil RBC values and were retained as COPCs for surface soil inside the landfill. Carbazole and chrysene were detected at concentrations less than their respective residential soil RBC values but were re-included as COPCs for subsurface soil inside the landfill because of the possible additive effect of cPAHs.

4,4'-DDT and heptachlor epoxide were detected at concentrations above their respective residential soil RBC values and were retained as COPCs for subsurface soil inside the landfill.

Aroclor-1260 was detected in the subsurface soil inside the landfill at Site 1 at a maximum concentration exceeding its residential soil RBC value and was retained as a COPC for surface soil inside the landfill.

2,4,6-Trinitrotoluene; 2,4-dinitrotoluene; 2-Amino-4,6-dinitrotoluene; 4-amino-2,6-dinitrotoluene; and RDX were detected at concentrations above their respective residential soil RBC values and were retained as COPCs for subsurface soil inside the landfill.

Inorganics were detected in all subsurface soil samples collected inside the landfill at Site 1. Aluminum, antimony, arsenic, barium, cadmium, chromium, copper, iron, lead, manganese, nickel, thallium, and zinc were detected at concentrations greater than residential soil RBC values and were retained as COPCs for subsurface soil inside the landfill. The results of the COPC selection for Site 1 subsurface soil inside the landfill are presented in Table 7-3.

#### 7.1.3.4 Subsurface Soil Outside Landfill

During the 2001 sampling event, five subsurface soil samples were collected outside the landfill from depths greater than one foot and analyzed for VOCs, SVOCs, pesticides, PCBs, nitramine compounds, and inorganics. During the 1992 Site Investigation at Site 1, two subsurface soil samples were collected outside the landfill from depths greater than one foot bgs and analyzed for VOCs, SVOCs, and inorganics. The COPC selection summary for the subsurface soil outside the landfill is presented on Table 7-4.

There were no VOCs detected in the subsurface soil outside the landfill at concentrations above corresponding residential soil RBC values. Therefore, VOCs were not retained as COPCs for subsurface soil outside the landfill.

Benzo(a)pyrene was detected at a concentration above its residential soil RBC value and was retained as a COPC for surface soil inside the landfill. Benzo(a)anthracene, benzo(b)fluoranthene, benzo(k)fluoranthene, carbazole, chrysene, dibenzo(a,h)anthracene, and indeno(1,2,3-cd)pyrene were detected at concentrations less than their respective residential soil RBC values but were re-included as COPCs for subsurface soil inside the landfill because of the possible additive effect of cPAHs.

There were no pesticides detected in the subsurface soil outside the landfill at concentrations above corresponding residential soil RBC values. Therefore, pesticides were not retained as COPCs for subsurface soil outside the landfill.

There were no PCBs detected in the subsurface soil outside the landfill at concentrations above corresponding residential soil RBC values. Therefore, PCBs were not retained as COPCs for subsurface soil outside the landfill.

Inorganics were detected in all subsurface soil samples collected outside the landfill at Site 1. Antimony, arsenic, chromium, iron, nickel, and thallium were detected at concentrations greater than residential soil RBC values and were retained as COPCs for subsurface soil outside the landfill. The results of the COPC selection for the subsurface soil outside the landfill are presented on Table 7-4.

#### 7.1.3.5 Groundwater

Six groundwater samples (five environmental and one duplicate samples) collected during the 2000 RI groundwater sampling effort were analyzed for VOCs, SVOCs, pesticides, PCBs, nitramine compounds, and unfiltered and filtered inorganics (metals). It should be noted that VOCs, pesticides, and PCBs were not detected in the groundwater collected at Site 1; therefore, these constituents were not retained as groundwater COPCs. The COPC selection summary for the groundwater is presented on Table 7-5.

Bis(2-ethylhexyl)phthalate was detected in the groundwater at a concentration greater than its tap water RBC concentration and was retained as a groundwater COPC.

4-Nitrotoluene and HMX were detected in the groundwater at concentrations below their respective tap water RBC values and were not retained as groundwater COPCs.

Total arsenic, iron, and manganese were detected at concentrations above their respective tap water RBC values and retained as total groundwater COPCs. Dissolved arsenic, cadmium, iron, and manganese exceeded their respective tap water RBC values and were retained as dissolved groundwater COPCs.

#### 7.1.3.6 Surface Water

Five surface water samples were collected during the 2001 sampling event at Site 1 and were analyzed for VOCs, SVOCs, pesticides, PCBs, nitramine compounds, and total inorganics. Four surface water samples were collected during the 1998 Field Investigation at Site 1 and were analyzed for VOCs, SVOCs, pesticides, PCBs, nitramine compounds, and total inorganics. However, VOCs, pesticides, PCBs, and nitramines were not detected in the surface water. The COPC selection summary for the surface water is presented on Table 7-6. Refer to Section 7.1.1 for an explanation of the screening criteria used for surface water COPC selection.

Methyl acetate was detected in the surface water at a concentration less than surface water screening criteria (i.e., tap water RBC value multiplied by a factor of ten [Baker, 2001]) and was not retained as a surface water COPC.

Bis(2-ethylhexyl)phthalate was detected in the surface water at concentrations less than surface water screening criteria and was not retained as a surface water COPC.

Inorganics were detected in all of the surface water samples collected. Arsenic was detected at concentrations that exceeded surface water screening criteria, and it was retained as a COPC for the surface water. The results of the COPC selection for the surface water are presented on Table 7-6.

#### 7.1.3.7 Sediment

Twenty-four sediment samples were collected during the 2001 sampling event and were analyzed for VOCs, SVOCs, pesticides, PCBs, nitramine compounds, and inorganics. Five sediment samples (four environmental and one duplicate samples) were collected during the 1998 Field Investigation at Site 1 and were analyzed for VOCs, SVOCs, pesticides, PCBs, nitramine compounds, and inorganics. However, nitramines were not detected in the sediment. The COPC selection summary for the sediment is presented on Table 7-7.

There were no VOCs detected in the sediment at concentrations above sediment screening values (USEPA Region III residential soil RBC values). Therefore, VOCs were not retained as sediment COPCs.

Benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, dibenz(a,h) anthracene, and indeno(1,2,3-cd)pyrene were detected in the sediment at concentrations that exceeded their respective sediment screening values and were retained as sediment COPCs. Benzo(k)fluoranthene, carbazole, and chrysene and were detected at concentrations less than their sediment screening values but were re-included as COPCs for sediment because of the possible additive effect of cPAHs.

There were no PCBs detected at concentrations above corresponding sediment screening values and therefore, were not retained as sediment COPCs.

Aroclor-1254 and Aroclor-1260 were detected at maximum concentrations that exceeded corresponding sediment screening values and were retained as sediment COPCs.

Inorganics were detected in all of the sediment samples collected. Aluminum, antimony, arsenic, chromium, copper, iron, lead, manganese, and nickel were retained as sediment COPCs because they exceeded sediment screening values. The results of the COPC selection for the sediment are presented on Table 7-7.

#### 7.1.4 Summary of COPCs

The following presents a comprehensive list of all selected COPCs, by medium, identified at Site 1.

**Surface soil inside the landfill:** Benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, carbazole, chrysene, dibenz(a,h)anthracene, indeno(1,2,3-cd)pyrene, phenanthrene, 4,4'-DDT, dieldrin, Aroclor-1248, Aroclor-1260, aluminum, antimony, arsenic, cadmium, chromium, copper, iron, lead, manganese, nickel, thallium, and zinc.

**Surface soil outside the landfill:** Benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, carbazole, chrysene, dibenz(a,h)anthracene, indeno(1,2,3-cd)pyrene, arsenic, chromium, copper, iron, manganese, and thallium.

**Subsurface soil inside the landfill:** Benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, carbazole, chrysene, dibenz(a,h)anthracene, indeno(1,2,3-cd)pyrene, 4,4'-DDT, Aroclor-1260, heptachlor epoxide, 2,4,6-Trinitrotoluene, 2,4-dinitrotoluene, 2-amino-4,6-

dinitrotoluene, 4-amino-2,6-dinitrotoluene, RDX, aluminum, antimony, arsenic, barium, cadmium, chromium, copper, iron, lead, manganese, nickel, thallium, and zinc.

**Subsurface soil outside the landfill:** Benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, carbazole, chrysene, dibenz(a,h)anthracene, indeno(1,2,3-cd)pyrene, antimony, arsenic, chromium, iron, nickel, and thallium.

**Groundwater:** Bis(2-ethylhexyl)phthalate, total arsenic, total iron, total manganese, dissolved arsenic, dissolved cadmium, dissolved iron, and dissolved manganese.

**Surface water:** Arsenic.

**Sediment:** Benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, carbazole, chrysene, dibenz(a,h)anthracene, indeno(1,2,3-cd)pyrene, Aroclor-1254, Aroclor-1260, aluminum, antimony, arsenic, chromium, copper, iron, lead, manganese, and nickel.

## 7.2 Exposure Assessment 6.2 Exposure Assessment

The exposure assessment addresses each potential current and future exposure pathway in soil, groundwater, surface water, sediment, and air. To determine whether human exposure could occur at Site 1 in the absence of remedial action, an exposure assessment that identified potential exposure pathways and receptors was conducted. The following four elements were considered to ascertain whether a complete exposure pathway exists (USEPA, 1989):

- A source and potential mechanism of chemical release
- An environmental retention or transport medium
- A point of potential human contact with the contaminated medium
- An exposure route (e.g., ingestion) at the contact point

Relevant equations for assessing intakes and exposure factors were obtained from RAGS (USEPA, 1989), Exposure Factors Handbook (USEPA, 1997), Dermal Exposure Assessment: Principles and Applications, Interim Report (USEPA, 1992a), and Standard Default Exposure Factors, Interim Final (USEPA, 1991). Unless otherwise noted, the values assumed for exposure factors used in the dose evaluation equations for assessing exposure were obtained

from recommended values from statistical distributions provided in the Exposure Factors Handbook and other EPA HHRA guidance documents.

### **7.2.1 Land Use and Potential Human Receptors**

CAX will most likely continue to function as a storage depot. Today the mission of CAX includes supplying Atlantic Fleet ships and providing recreational opportunities to military and civilian personnel. Outdoor recreational facilities and activities available include: 13 cabins, 19 recreational vehicle sites, camp sites, an 18-hole golf course, swimming pool, ball fields, freshwater and saltwater fishing, boating, wildlife watching, and hunting (Department of the Navy [DON], 1998). There are no drinking water wells at CAX. Drinking water is supplied by the treatment plant that draws surface water from Jones Pond, which is located on CAX property south of the Colonial National Historic Parkway.

Based on information available regarding the physical features, site setting, site historical activities, and current and expected land uses, six potential human receptors have been selected for evaluation both inside and outside the Site 1 landfill boundaries.

- Current On-Site Adolescent Trespassers (7-15 years)
- Current On-Site Adult Trespassers
- Current/Future On-Site Adult Industrial/Commercial Workers
- Future Resident Younger Children (1-6 years)
- Future Resident Adults
- Future Adult Construction Workers

Current potential human receptors potentially exposed to COPCs detected in environmental media at Site 1 are limited to on-site adult and adolescent trespassers and industrial/commercial workers. Although future residential development of Site 1 is highly unlikely, future residential exposure for potential adult and younger child receptors was considered in accordance with USEPA guidance and also to provide an upper bound for potential risks. To evaluate potential exposure to groundwater, younger child and adult residents were considered to be potentially exposed to organic and total inorganic COPCs in the Yorktown Eastover aquifer using a nonpotable, beneficial use scenario at the site; since the shallow aquifer system within CAX is not used as a potable water source. In addition, future construction workers that may perform excavation and housing construction activities, were evaluated as potential receptors.

## 7.2.2 Exposure Pathways (tc \13 "6.2.1 Potential Human Receptors)

The potential human receptors and exposure pathways evaluated at Site 1 were selected considering current and future potential land use for CAX. The following paragraphs present the rationale for the selection of each pathway.

Data from soil samples collected inside the landfill were segregated from data from soil samples collected outside the landfill for evaluation in this human health risk assessment. This was done to assess potential exposures from soils within the landfill versus exposures from soils outside the landfill. Therefore, potential exposures to the above referenced human receptors were evaluated under two separate exposure scenarios: one exposure scenario including soil *inside* the landfill (i.e., landfill material) and another exposure scenario including soil *outside* the landfill (i.e., soil cover or native soil). Data from samples collected for groundwater (future receptors only), surface water, and sediment were not separated in this manner, but were evaluated as whole units by media (e.g., all groundwater data evaluated as one data set).

Potential exposure to COPCs and media of concern for the current on-site adult and adolescent trespassers includes accidental ingestion and dermal contact with the surface soil (inside and outside the landfill), surface water, and sediment. Fugitive dust generation from surface soil is not considered to be a significant potential release mechanism at this site since it is covered to a great extent by vegetation. However, inhalation of fugitive dusts from surface soil (inside and outside the landfill) is evaluated to maintain a conservative approach. Inhalation is not considered a significant pathway for surface water or sediment exposure and is not evaluated in this HHRA.

In a current scenario, on-site industrial/commercial workers include CAX personnel who would be responsible for groundskeeping activities. In the future scenario, the commercial/industrial worker would include any employee of the facility. Potential exposure to COPCs and media of concern at Site 1 for the current industrial/commercial worker includes accidental ingestion and dermal contact with surface soil (inside and outside the landfill), inhalation of fugitive dusts emanating from surface soil disturbed during maintenance activities, as well as accidental ingestion and dermal contact with surface water and sediment.

Despite the unlikely nature of residential development by the military or general public, future residential exposure by children and adults will be evaluated. Future adult and child residential

receptors could potentially be exposed to COPCs in surface soil (inside and outside the landfill), subsurface soil (inside and outside the landfill), dissolved groundwater (nonpotable, beneficial-use scenario), surface water, and sediment, by ingestion, dermal contact, and inhalation of fugitive dusts from soil. In this HHRA, it is assumed that subsequent to excavation for residential development, subsurface soil would be brought to the surface and is therefore, considered an exposure pathway for future residents. As with the on-site trespasser receptor, fugitive dust generation from surface soil is not considered to be a significant potential release mechanism. However, it is evaluated in this HHRA to maintain a conservative approach. Also, inhalation is not considered a significant pathway for surface water or sediment exposure and is not evaluated in this HHRA.

Potential exposure to COPCs at Site 1 could occur in the future if utilities or buildings in the area are constructed. The future construction worker will, therefore, be evaluated for accidental ingestion, dermal contact, and inhalation of fugitive dust in surface and subsurface soil (inside and outside the landfill) during excavation activities. Contact with the water-bearing zone is also possible during construction activities. Therefore, the future construction workers are also evaluated for accidental ingestion and dermal contact with total groundwater.

In summary, the following potential human exposure receptors and exposure pathways are being retained for quantitative evaluation in this HHRA.

- Current On-Site Adult and Adolescent (7-15 years old) Trespassers:
  - Accidental ingestion of surface soil (inside and outside the landfill)
  - Dermal contact with surface soil (inside and outside the landfill)
  - Inhalation of fugitive dust in surface soil (inside and outside the landfill)
  - Accidental ingestion of surface water
  - Dermal contact with surface water
  - Accidental ingestion of sediment
  - Dermal contact with sediment
  
- Current/Future On-Site Adult Industrial/Commercial Workers:
  - Accidental ingestion of surface soil (inside and outside the landfill)
  - Dermal contact with surface soil (inside and outside the landfill)
  - Inhalation of fugitive dust in surface soil (inside and outside the landfill)
  - Accidental ingestion of surface water

- Dermal contact with surface water
- Accidental ingestion of sediment
- Dermal contact with sediment
  
- Future On-Site Adult and Younger Child (1-6 years old) Residents:
  - Accidental ingestion of surface soil (inside and outside the landfill)
  - Dermal contact with surface soil (inside and outside the landfill)
  - Inhalation of fugitive dust in surface soil (inside and outside the landfill)
  - Accidental ingestion of subsurface soil (inside and outside the landfill)
  - Dermal contact with subsurface soil (inside and outside the landfill)
  - Inhalation of fugitive dust in subsurface soil (inside and outside the landfill)
  - Ingestion of groundwater during beneficial use
  - Dermal contact with groundwater during beneficial use
  - Accidental ingestion of surface water
  - Dermal contact with surface water
  - Accidental ingestion of sediment
  - Dermal contact with sediment
  
- Future On-Site Adult Construction Workers:
  - Accidental ingestion of surface soil (inside and outside the landfill)
  - Dermal contact with surface soil (inside and outside the landfill)
  - Inhalation of fugitive dust in surface soil (inside and outside the landfill)
  - Accidental ingestion of subsurface soil (inside and outside the landfill)
  - Dermal contact with subsurface soil (inside and outside the landfill)
  - Inhalation of fugitive dust in subsurface soil (inside and outside the landfill)

### **7.2.3 Conceptual Site Model**

Development of a conceptual site model of potential exposure is critical in evaluating all potential exposures for the aforementioned human receptors. The conceptual site model describes the potential exposure in terms of potential sources of contamination, affected media, and all potential routes of migration of the contaminants present. The conceptual site model for Site 1 is presented in Figure 7-1.

#### 7.2.4 Quantification of Exposure

The chemical concentrations used in the estimation of Chronic Daily Intakes (CDIs) and Dermal Absorbed Doses (DADs) for each medium are considered to be representative of potential exposure encountered by each receptor throughout the time of exposure. Groundwater and surface water are in constant motion; thus, chemical concentrations detected in these media may be variable over time. Soil and sediment generally move more slowly by erosion. Therefore, groundwater and surface water contaminant concentrations may be best represented by the most recent data, while using all of the chemical data collected over time may be more representative for soil and sediment. The manner in which environmental data are represented also depends on the number of samples and sampling locations available for a given area and a given medium. For example, exposure can occur to a portion of the site (i.e., a "hotspot") or the entire site depending on the type of scenario considered for a given receptor.

To quantify exposure, analytical data must be evaluated to determine its distributional nature. In general, two types of distributions are applied to environmental data; these are the normal and lognormal distributions. For example, most large data sets from soil sampling are lognormally distributed rather than normally distributed. The geometric mean is the best estimator of central tendency for a lognormal data set (USEPA, 1992c). However, most Agency health criteria are based on the long-term average exposure, which is expressed as the sum of all daily intakes divided by the total number of days in the averaging period. The geometric mean of a set of sampling results may not adequately represent random exposure and the cumulative intake that would result from long-term contact with site contaminants.

Potential exposure to soil, surface water, and sediment at Site 1, regardless of location, is considered as having an equal probability of occurrence as an individual moves randomly across the site. Therefore, for these media, the exposure point concentration for a constituent in the intake equation can be reasonably estimated as the arithmetic average concentration of site sampling data. USEPA supplemental risk assessment guidance (USEPA, 1992c) states that the average concentration is an appropriate estimator of the exposure concentration for two reasons: 1) carcinogenic and chronic noncarcinogenic toxicity criteria are based on lifetime average exposures; and 2) the average concentration is most representative of the concentration that would be contacted over time. However, uncertainty is inherent in the estimation of the true average constituent concentration at the site.

A conservative estimate of the arithmetic average concentration recommended by USEPA (1992c) is the 95 percent upper confidence limit of the arithmetic mean concentration (95 percent Upper Confidence Limit [UCL]). In order to estimate the 95 percent UCL for soil, surface water, and sediment data sets, a normal distribution was assumed to represent the occurrence of all COPC-detected concentrations for sample data sets greater than or equal to five. The maximum concentration is used for the concentration term when the sample data set is five or less. Furthermore, if the 95 percent UCL of the arithmetic mean exceeds the maximum detected concentration in a given data set, the maximum detected concentration will be used to represent the concentration term for that COPC.

The 95 percent UCL was calculated using the following equation (USEPA, 1992c):

$$\text{Normal 95\% UCL} = \bar{x} + t(s/\sqrt{n})$$

Where:

Normal 95% UCL = 95<sup>th</sup> percent upper confidence limit for the arithmetic mean concentration

$\bar{x}$  = mean

s = standard deviation

t = Student t statistic (Gilbert, 1987)

n = number of samples

The maximum detected concentration was used to represent the exposure point concentration for COPCs selected for groundwater. This is because there was no evidence of an organic plume, and most of groundwater COPCs (both total and dissolved) were inorganics. Also, the groundwater data set is small (n = 6). Therefore, the maximum detected concentration was used to maintain a conservative approach. Since the maximum detected concentration of the COPCs was selected for the exposure point concentration, all wells were included in the data set (including the upgradient well, GW05).

Frequency of detection as well as maximum detected values for the analytical results are presented in Section 5.0 of this report. Ninety-five percent UCL values and mean values, derived for COPCs in all media at Site 1 are presented in Appendix J. The equations for estimating

intakes due to direct exposures to site-related chemicals for the various identified pathways are presented in Appendix K.

For results reported as "nondetect" (i.e., results flagged with the following validation qualifiers: U, UJ, UL, and UK), a value of one half of the sample-specific detection limit was used to calculate the 95 percent UCL. A value of half the detection limit was assigned to nondetects when estimating the 95 percent UCL and the arithmetic mean because the actual value could be between zero and a value just below the detection limit. Ninety-five percent UCLs were calculated only for the constituents detected in at least one sample collected from the environmental medium of interest. Estimated concentrations also were used to calculate the 95 percent UCL, such as "J"-qualified (estimated), "L"-qualified (estimated, biased low) and "K"-qualified (estimated, biased high) data. Reported concentrations qualified with an "R" (rejected) were not used in the statistical evaluation.

According to the Region III Modifications to the National Functional Guidelines (NFGs), reported organic and inorganic concentrations that were qualified with a "B" were evaluated against the available field and laboratory blanks. This qualifies the organic/inorganic as a nondetect due to laboratory contamination. For constituents considered by RAGS to be common laboratory contaminants, chemicals were deemed positive detects only if their concentration exceeded 10 times the maximum blank concentration. For constituents not considered common laboratory contaminants, chemicals were considered as positive detects only if their concentration exceeded 5 times the maximum blank concentration.

#### **7.2.5 Exposure Factors {tc \13 "6.2.4 Exposure Factors Used To Derive Chronic Daily Intakes}**

Table 7-8 presents the exposure factors used in the estimation of potential CDIs/DADs for COPCs retained for each receptor identified below. These exposure factors are input parameters in the dose evaluation equations that were used for both the reasonable maximum exposure (RME) and central tendency (CT) exposure scenarios. The goal of RME is to combine upper bound and mid-range exposure factors so that the result represents an exposure scenario that is both protective and reasonable, not the worst possible case (USEPA, 1991). The CT scenario uses some less conservative exposure factors to provide risk managers additional information to make risk-based decisions. In this HHRA, the CT scenario was evaluated only for the future residential receptor scenario. The same concentration term is used for both the RME and CT

scenarios. USEPA recommended exposure factors are used in conjunction with USEPA standard default exposure factors for both the CT and RME exposure scenarios. Furthermore, when USEPA exposure factors are not available, best professional judgment and site-specific information are used to derive a conservative and defensible value. The following paragraphs present the rationale for the selection of exposure factors for each receptor group evaluated in the baseline HHRA.

#### 7.2.5.1 Current On-Site Adult and Adolescent (7-15 years old) Trespassers

This scenario assumes that current adult and adolescent (7 - 15 years) trespassers could come into contact with surface soil, surface water, and sediment. Therefore, these receptors were evaluated for potential exposure to these media via incidental ingestion and dermal contact, and surface soil via inhalation of fugitive dust. A summary of the exposure parameters is discussed in the following paragraphs and presented on Table 7-8.

A 70 kg adult and a 45 kg adolescent (USEPA, 1997) were assumed to have exposure durations of 24 years and 9 years, respectively (USEPA, 1991). The exposure time in relationship to inhalation of fugitive dusts was estimated to be 12 hours per day (USEPA, 1997). The ingestion rate of surface soil and sediment was assumed to be 100 mg/day for both the adolescent and the adult (USEPA, 1989), with a 100 percent fraction ingested from the source (professional judgement). The IR for exposure to surface water for both the adult and adolescent trespassers was assumed to be 0.05 L/hour (USEPA, 1989), over an exposure time (ET) of 1 hour/day (USEPA, 1997). The exposure frequency was assumed to be 52 events/year, based on anticipated exposures of one day/week/year (professional judgement). A respiration rate of 0.83 m<sup>3</sup>/hr for the adult and adolescent (USEPA, 1991) was also used in relationship to inhalation of fugitive dusts from surface soil. Averaging times of 8,760 days for adults and 3,285 days for adolescents for noncarcinogens, and 25,550 days for carcinogens were also used (USEPA, 1989).

The USEPA recommended weighted SAF of 0.07 mg/cm<sup>2</sup> for the residential adult was used for the adult trespasser (USEPA, 1997a). This is based on the 50<sup>th</sup> percentile weighted SAF for gardeners, which is the activity determined to represent a reasonable, high-end contact activity. The USEPA recommended weighted 0.2 mg/cm<sup>2</sup> SAF for the young child was conservatively used for the adolescent trespasser and is based on the 95<sup>th</sup> percentile weighted SAF for children playing at a day care center or in wet soil (USEPA, 1997a). Dermal absorption values provided in USEPA RAGS Part E (USEPA, 2001) or USEPA Region III default dermal absorption values of

0.05 percent for VOCs, 10 percent for PAHs, 50 percent for SVOCs and nitramines, 6 percent for PCBs and pesticides, 1 percent for inorganics and 3.2 percent for arsenic were applied in evaluating dermal exposures to soil (USEPA, 1995d). Skin surface areas of 5,300 cm<sup>2</sup> for the adolescent (representing 50<sup>th</sup> percentile body-part SAF [average of male/female] for the head, forearms, hands, lower legs and feet of a <7 to <18 year old) and 5,700 cm<sup>2</sup> for the adult (USEPA 1997a) were assumed for the surface soil, surface water, and sediment scenarios.

#### 7.2.5.2 Current/Future On-Site Adult Industrial/Commercial Workers

This scenario assumes that current adult industrial/commercial workers could come into contact with surface soil, surface water, and sediment at Site 84. Therefore, this receptor was evaluated for potential exposure to surface soil, surface water, and sediment via incidental ingestion and dermal contact, and surface soil via inhalation of fugitive dust. A summary of the exposure parameters is discussed in the following paragraphs and presented on Table 7-8.

The IR for industrial/commercial workers exposed to surface soil and sediment was assumed to be 50 mg/day (USEPA, 1991), and the fraction ingested was assumed to be 100 percent. The IR for exposure to surface water was assumed to be 0.05 L/hour (USEPA, 1989), over an exposure time (ET) of 1 hour/day (USEPA, 1997). An exposure frequency (EF) of 250 days per year (USEPA, 1991) was used in conjunction with an exposure duration of 25 years (USEPA, 1991). A respiration rate of 0.83 m<sup>3</sup>/hr for the adult (USEPA, 1989) was also used in relationship to inhalation of fugitive dusts from surface soil. An averaging time (AT) of 70 years or 25,550 days was used for exposure to potentially carcinogenic compounds while an averaging time of 9,125 days was used for noncarcinogenic exposures.

There is a potential for industrial/commercial workers to absorb COPCs by dermal contact. A skin surface area of 3,300 cm<sup>2</sup> for an adult (USEPA, 1997) assumed to wear a short-sleeved shirt, long pants, and shoes, was used to evaluate dermal contact with soil, sediment, and surface water. The soil to skin adherence factor (SAF) of 0.2 mg/cm<sup>2</sup> was used and is based on the 50<sup>th</sup> percentile weighted SAF for utility workers, which is the activity determined by USEPA to represent a reasonable, high-end contact activity (USEPA, 1997). Dermal absorption values provided in USEPA RAGS Part E (USEPA, 2001) or USEPA Region III default dermal absorption values of 0.05 percent for VOCs, 10 percent for PAHs, 50 percent for SVOCs and nitramines, 6 percent for PCBs and pesticides, 1 percent for inorganics and 3.2 percent for arsenic were applied in evaluating dermal exposures to soil (USEPA, 1995d).

### 7.2.5.3 Future On-Site Adult and Younger Child (1-6 years old) Residents

This scenario assumes that future adult and young child (1-6 years) residents could come into contact with surface soil (inside and outside the Landfill), surface water, and sediment at Site 1. It is also conservatively assumed that the groundwater would occur via a nonpotable, beneficial use scenario. These receptors were evaluated for potential exposure to surface soil, surface water, and sediment via accidental ingestion and dermal contact, and surface soil via inhalation of fugitive dust. Future residents were also evaluated for potential exposure to groundwater via ingestion and dermal contact. Under the beneficial groundwater use scenario, exposures to organic and total inorganic COPCs were evaluated. A summary of the exposure parameters is discussed in the following paragraphs and presented on Table 7-8. Unless otherwise noted, the Central Tendency (CT) exposure parameters are the same as for Reasonable Maximum Exposure (RME).

Future adult and young child residents could contact surface soil, surface water, and sediment during outdoor recreational activities in the area immediately surrounding their homes. A 70 kg adult and a 15 kg child were assumed for exposure durations of 24 years and 6 years, respectively (USEPA, 1989). Exposure durations of 9 years for the adult and 6 years for the child were used for CT exposure (USEPA, 1993d). The exposure time was conservatively assumed to be 24 hours per day. The ingestion rate for surface soil and sediment was assumed to be 200 mg/day for the young child and 100 mg/day for the adult (USEPA, 1991), with a 100 percent fraction ingested from source, over 350 days/year (USEPA, 1991) for soil and 40 days/year (professional judgement) for surface water and sediment. Ingestion rates for surface soil and sediment of 100 mg/day for the young child and 50 mg/day for the adult over 234 days per year were used for CT exposure (USEPA, 1993d). Respiration rates of 0.83 m<sup>3</sup>/hr for the child (professional judgement) and 0.83 m<sup>3</sup>/hr for the adult (USEPA, 1991) were also used in relationship to inhalation of fugitive dusts. Averaging times of 8,760 days for adults and 2,190 days for children for noncarcinogens, and 25,550 days for carcinogens were also used (USEPA, 1989).

The USEPA recommended weighted SAF of 0.07 mg/cm<sup>2</sup> was used for the residential adult (USEPA, 1997). This is based on the 50<sup>th</sup> percentile weighted SAF for gardeners, which is the activity determined to represent a reasonable, high-end contact activity. The USEPA recommended weighted 0.2 mg/cm<sup>2</sup> SAF for the young child was used and is based on the 95<sup>th</sup> percentile weighted SAF for children playing at a day care center or in wet soil (USEPA, 1997).

Dermal absorption values provided in USEPA RAGS Part E (USEPA, 2001) or USEPA Region III default dermal absorption values of 0.05 percent for VOCs, 10 percent for PAHs, 50 percent for SVOCs and nitramines, 6 percent for PCBs and pesticides, 1 percent for inorganics and 3.2 percent for arsenic were applied in evaluating dermal exposures to soil (USEPA, 1995d). Skin surface areas of 2,800 cm<sup>2</sup> for the young child and 5,700 cm<sup>2</sup> for the adult (USEPA 1997) were assumed for the surface soil, surface water, and sediment scenario. These are the SA values currently recommended by the USEPA for exposure to contaminated soil and are the averages of the 50<sup>th</sup> percentiles for males and females greater than 18 years of age (adults) and from <1 to <6 years old (young children). Skin surface areas of 2,000 cm<sup>2</sup> for the young child and 5,000 cm<sup>2</sup> for the adult (USEPA, 1997) were assumed for the CT exposure scenario.

The adult and child residents were evaluated for ingestion and dermal exposures to groundwater used for non-potable purposes. Most of the same assumptions used for estimating exposures to surface soil (i.e., surface areas, body weight, and noncarcinogenic and carcinogenic averaging times) were also applied to the evaluation of ingestion and dermal exposures to shallow groundwater during beneficial use activities. However, a groundwater ingestion rate of 0.05 L/day was used to evaluate both the older child and adult. Since no other ingestion rates have been published in the literature for such a scenario, this ingestion rate was used as a conservative estimate that was derived from professional judgement based on USEPA's (1989) surface water ingestion rate of 0.05 L/hour established for a swimming scenario. The surface water ingestion value of 0.05 L/hour was adjusted to 0.05 L/day, thereby yielding a conservative value that is more reflective of the beneficial use scenario being evaluated (i.e., incidental exposure while watering lawns and washing cars), than is the value for the swimming scenario. An exposure time (ET) of 1 hour/day (USEPA, 1997) was used. The exposure frequency was assumed to be 40 days/year (professional judgement). Equations and estimated, chemical-specific K<sub>p</sub> values presented by USEPA (USEPA, 1992a) were used to estimate the absorption of organic COPCs by skin exposed to shallow groundwater.

#### 7.2.5.4 Future On-Site Adult Construction Workers

Potential exposures to soil COPCs may occur to construction workers while performing soil excavation and construction activities at Site 1. Exposure pathways evaluated include accidental ingestion, dermal contact, and inhalation of fugitive dust of surface and shallow subsurface soil (inside and outside the Landfill). Accidental ingestion and dermal contact with groundwater were also evaluated as potential exposure pathways. A summary of the exposure parameters is

discussed in the following paragraphs and presented on Table 7-8. Exposure was assumed to occur for 8 hours per day (professional judgement), 250 days per year (USEPA, 1991), for a construction period of 1 year (professional judgement). A USEPA default value for the soil ingestion rate of 480 mg/day, a 100 percent fraction ingested from source, and a respiration rate of 0.83 m<sup>3</sup>/hour (USEPA, 1991) were also assumed for a 70 kg construction worker (USEPA, 1989). An ingestion rate of 0.05 L/day (professional judgement) was used for accidental ingestion of groundwater. A skin surface area of 3,300 cm<sup>2</sup> for an adult (USEPA 1997) assumed to wear a short-sleeved shirt, long pants, and shoes, was used to evaluate dermal contact with soil and groundwater. The soil to skin adherence factor of 0.2 mg/cm<sup>2</sup> (USEPA, 1997) was used. Dermal absorption values provided in USEPA RAGS Part E (USEPA, 2001) or USEPA Region III default dermal absorption values of 0.05 percent for VOCs, 10 percent for PAHs, 50 percent for SVOCs and nitramines, 6 percent for PCBs and pesticides, 1 percent for inorganics and 3.2 percent for arsenic were applied in evaluating dermal exposures to soil (USEPA, 1995d). The averaging time of 365 for noncarcinogens and 25,550 days for carcinogens, respectively, were also used (USEPA, 1989).

### **7.3 Toxicity Assessment**

Section 7.2 presented potential exposure pathways and receptors for this baseline HHRA. This section will review the available toxicological information for COPCs retained for quantitative evaluation.

An important component of the HHRA process is the relationship between the dose of a compound (amount to which an individual or population is potentially exposed) and the potential for adverse health effects resulting from exposure to that dose. Dose-response relationships provide a means by which potential public health impacts may be evaluated. Standard Reference Doses (RfDs) and/or Carcinogenic Slope Factors (CSFs) have been developed for many of the COPCs. This section provides a brief description of these parameters.

#### **7.3.1 Reference Doses**

The RfDs and Reference Concentrations (RfCs for inhalation) are developed for chronic and/or subchronic human exposure to chemicals and are based solely on the noncarcinogenic effects of chemical substances. These values are defined as an estimate of a daily exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable

risk of adverse effects during a lifetime. The RfD is expressed as dose (mg) per unit body weight (kg) per unit time (day). The RfC is expressed as dose (mg) per cubic meter of air (m<sup>3</sup>).

### 7.3.2 Carcinogenic Slope Factors

CSFs are used to estimate an upper bound lifetime probability of an individual developing cancer as a result of exposure to a particular level of a potential carcinogen (USEPA, 1989). This factor is reported in units of (milligram per kilogram [mg/kg/day])<sup>-1</sup> and is derived through an assumed low-dosage linear multistage model and an extrapolation from high to low dose-responses determined from animal studies. The value used in reporting the slope factor is the 95 percent UCL. CSFs can also be derived from USEPA promulgated unit risk values for air and/or water. CSFs derived from unit risks cannot, however, be applied to environmental media other than the medium considered in the unit risk estimate.

Slope factors are also accompanied by weight-of-evidence classifications, which designate the strength of the evidence that the COPC is a potential human carcinogen.

Quantitative indices of toxicity and USEPA weight-of-evidence classifications are presented in Table 7-9 for the identified COPCs. The hierarchy (USEPA, 1989) for choosing these values was:

- Integrated Risk Information System (IRIS) (USEPA, 2002)
- Health Effects Assessment Summary Table (HEAST) (USEPA, 1997a)
- National Center for Environmental Assessment (NCEA)

The IRIS database is updated monthly and contains both verified RfDs, RfCs and CSFs. The USEPA has formed an RfD work group to review existing data used to derive RfDs and RfCs. Once this task has been completed the verified RfD appears in IRIS. Like the RfD Work Group, the USEPA has also formed the Carcinogen Risk Assessment Verification Endeavor (CRAVE) Work group to review and validate toxicity values used in developing CSFs. Once the slope factors have been verified via extensive peer review, they also appear in the IRIS data base.

HEAST, on the other hand, provides both interim (unverified) and verified RfDs, RfCs and CSFs. This document is published quarterly and incorporates any applicable changes to its data base.

### **7.3.3 Dermal Absorption Efficiency**

#### **{tc \13 "6.3.3 Dermal Absorption Efficiency}**

Many of the RfDs and CSFs are derived from oral toxicological studies based on administered dose, and do not account for the amount of a substance that can penetrate exchange boundaries after contact (e.g., absorbed dose). As a result, there is very little information available regarding dermal toxicity criteria. Therefore, in order to account for a difference in toxicity between an administered dose and an absorbed dose, the RfDs and CSFs (that were based on an administered dose) were adjusted, as described by the USEPA (USEPA, 1989), using experimentally-derived oral absorption efficiencies. The adjustment for the oral RfD that would correspond to a dermally absorbed dose is represented by multiplying the RfD by an oral absorption efficiency. The adjustment for the oral CSF that would correspond to the dermally absorbed dose is represented by dividing the CSF by an oral absorption efficiency. The oral absorption efficiencies were obtained from sources such as the NCEA, IRIS, Agency for Toxic Substance and Disease Registry (ATSDR) toxicological profiles, toxicology publications, toxicology references, and USEPA Regional Offices. RAGS, Appendix A was used as the primary resource for oral absorption efficiencies. The most recent update to these oral absorption efficiencies was in April 1999. In some instances, published information was not available to determine the absorption efficiency, or published information indicated that the absorption efficiency was low for both dermal and oral routes of exposure (i.e., chromium). On these occasions, adjustments to the toxicity value were not conducted (e.g., an absorption efficiency of 100 percent was assumed).

It should be noted that PAH compounds were not evaluated for dermal contact and absorption in the risk assessment. Studies have shown that effects due to absorption of PAHs may not be significant since adverse effects are typically observed at the point of application; therefore, USEPA Region III has adopted the non-evaluation approach for PAH exposures via the dermal pathway.

### **7.3.4 Lead**

Lead was identified as a COPC in samples collected from surface soil and subsurface soil inside the landfill at Site 1. Currently, health-based criteria are not available for evaluating either the noncarcinogenic or carcinogenic effects of lead exposure. The USEPA has not developed health-based criteria because a threshold level for many noncancer health effects has not been identified in infants and younger children (i.e., the most sensitive populations). Consequently, risk from lead exposure was not calculated for the site.

To evaluate lead at sites, the USEPA developed an Integrated Exposure Uptake Biokinetic (IEUBK) Model for Lead in Children, Windows™ version (USEPA, 2001). This model utilizes site-specific exposure parameters to estimate blood lead levels in infants and young children. The USEPA considers remediation necessary if a five percent probability or greater exists that the predicted child blood level will exceed 10 micrograms per deciliter (µg/dl) as a result of contact with lead-containing media at the site.

There are several criteria available for lead level comparisons in the form of standards and/or criteria. These standards/criteria include federal and state Maximum Contaminant Levels (MCLs). In addition, there is an Office of Solid Waste and Emergency Response (OSWER) action level for lead of 400 mg/kg in residential soil. At Site 1, the maximum concentrations of lead found in surface soil and subsurface soil inside the landfill exceeded the OSWER action level for residential soil. Consequently, the lead IEUBK model was utilized to evaluate the risk associated with exposure to lead in the surface and subsurface soil inside the landfill. The normal 95 percent UCLs for lead in surface soil and subsurface soil were used as exposure point concentrations for the IEUBK model. All other exposure parameters used in the model were default values recommended by the IEUBK model guidance document (USEPA, 2001).

#### **7.4 Risk Characterization**

The risk characterization combines the selected COPCs, the exposure assessment, and the toxicity assessment to produce a quantitative estimate of current potential human health risks associated with Site 1. Estimated ILCRs and Hazard Indexes (HIs) for the identified potential current adult and adolescent on-site trespassers who could be exposed to COPCs via dermal contact, accidental ingestion, and inhalation of fugitive dust in surface soil, and dermal contact and accidental ingestion of surface water and sediment; current/future industrial/commercial worker who could be exposed to COPCs via dermal contact, accidental ingestion, and inhalation of fugitive dust in surface soil and dermal contact and accidental ingestion of surface water and sediment; the future adult and child residents who could be exposed to COPCs via dermal contact, ingestion, and inhalation of fugitive dust in surface soil, dermal contact and accidental ingestion of surface water and sediment, and groundwater (adult and younger child under a nonpotable use scenario); and the future adult construction worker who could be exposed to COPCs via dermal contact, accidental ingestion, and inhalation of fugitive dust in surface soil and subsurface soil, as well as dermal contact and accidental ingestion of total groundwater are discussed in this section. The

ILCRs and HIs were calculated for each of the soil (inside and outside the landfill), groundwater, surface water, and sediment COPCs using the 95 percent UCL of the arithmetic mean as the exposure point concentration, or the maximum concentration if the 95 percent UCL exceeded the maximum.

#### 7.4.1 Carcinogenic Compounds

Quantitative risk calculations for potentially carcinogenic compounds estimate inferentially the potential ILCR for an individual in a specified population. This unit of risk refers to a potential cancer risk that is above the background cancer risk in unexposed individuals. For example, an ILCR of  $1 \times 10^{-6}$  indicates that an exposed individual has an increased probability of one in one million of developing cancer subsequent to exposure, over the course of their lifetime.

The potential lifetime ILCR for an individual was estimated from the following relationship:

$$\text{ILCR} = \sum_{i=1}^n (\text{CDI}_i \text{ or DAD}_i) \times \text{CSF}_i$$

where the  $\text{CSF}_i$  is expressed as  $(\text{mg/kg/day})^{-1}$  for compound I, and the  $\text{CDI}_i$  and Dermal Absorbed Dose ( $\text{DAD}_i$ ) are expressed as  $\text{mg/kg/day}$  for compound I. Since the units of  $\text{CSF}_i$  are  $(\text{mg chemical/kg body weight-day})^{-1}$  and the units of intake or dose are  $\text{mg chemical/kg body weight-day}$ , the ILCR value is dimensionless. The aforementioned equation was derived assuming that cancer is a nonthreshold process and that the potential excess risk level is proportional to the cumulative intake over a lifetime.

For quantitative estimation of risk, it is assumed that cancer risks from various exposure routes are additive. Estimated ILCR values will be compared to  $1 \times 10^{-6}$  to  $1 \times 10^{-4}$  which represents the target risk range of ILCR values considered by the USEPA to represent an acceptable (i.e., de minimis) risk (USEPA, 1990).

#### 7.4.2 Noncarcinogenic Compounds

Noncarcinogenic compounds assume that a threshold toxicological effect exists. Therefore, the potential for noncarcinogenic effects is calculated by comparing (i.e., dividing)  $CDI_i$  and  $DAD_i$  levels with  $RfD_i$  for each COPC.

Noncarcinogenic effects are estimated by calculating the HQ for individual chemicals and the HI for overall chemicals and pathways by the following equation:

$$HI = \sum_{i=1}^n HQ_i$$

where:  $HQ_i = (CDI_i \text{ or } DAD_i) / RfD_i \text{ or } RfC_i$

An HQ is the ratio of the daily intake or absorbed dose to the reference dose (or reference concentration for inhalation exposure).  $CDI_i$  is the chronic daily intake (mg/kg/day) of contaminant I;  $DAD_i$  is the dermally absorbed dose (mg/kg/day) of contaminant I, and  $RfD_i$  is the reference dose (mg/kg/day) of the contaminant I over a prolonged period of exposure.  $RfC_i$  is the reference concentration used when determining exposure due to inhalation. Since the units of  $RfD$  are mg/kg-day and the units of  $CDI/DAD$  are mg/kg-day, the HQ and HI are dimensionless. To account for the additivity of noncarcinogenic risk following exposure to numerous chemicals, the HI, which is the sum of all the HQs, will be calculated. A ratio of 1.0, considered by the USEPA to represent an acceptable hazard level (USEPA, 1990), is used for examination of the HQ and HI. Ratios less than 1.0 indicate that adverse noncarcinogenic health effects are unlikely. Ratios greater than 1.0 indicate the potential for adverse noncarcinogenic health effects to occur at that exposure level and caution should be exercised. However, this does not mean that adverse effects will definitely be observed since the  $RfD$  incorporates safety and modifying factors to ensure that it is well below that dose for which adverse effects have been observed. This procedure assumes that the risks from exposure to multiple chemicals are additive, an assumption that is probably valid for compounds that have the same target organ or cause the same toxic effect. It should be noted that this summation approach ignores potential interactions among the various chemicals at the site, which may either enhance or reduce the potential health effects.

### **7.4.3 Potential Human Health Effects**

Total risks were estimated by site for future residential receptors using both RME and CT exposure scenarios. Risks from surface soil were derived and presented from two perspectives: inside versus outside the landfill. Groundwater risks were estimated for the Yorktown Eastover aquifer under a nonpotable, beneficial use scenario. Current trespassers were evaluated for surface soil, surface water and sediment exposure. Future construction workers were evaluated for surface soil, subsurface soil, and groundwater exposures. Current/future industrial/commercial workers were evaluated for surface soil, surface water and sediment exposures.

The most significant carcinogenic and noncarcinogenic risks estimated were for the future resident adult and child receptors within the landfill. The soil ingestion exposure pathways inside the landfill contributed predominantly to these elevated risk levels. Table 7-10 presents the total site risks for potential current and future human exposures, respectively, to COPCs identified in environmental media at Site 1. Tables 7-11 through 7-15 present a break down of the total site risks by medium and pathway and provide target organ analysis for those HI values that exceeded USEPA acceptable level of 1.0. Exceedences of USEPA acceptable risk criteria are represented by the shaded regions of the tables. Risk calculations presenting the ILCR and HI values, by pathway and medium, for current adult and adolescent on-site trespassers, current on-site industrial/commercial workers, future on-site adult and child residents, and future adult construction workers at Site 1 are presented in Appendix K.

#### **7.4.3.1 Current On-Site Adult and Adolescent Trespassers**

There were no carcinogenic risks calculated that exceeded USEPA's acceptable risk range of  $1 \times 10^{-4}$  to  $1 \times 10^{-6}$  for the current on-site adult and adolescent trespassers upon exposure to surface soil (both inside and outside the landfill) via accidental ingestion, dermal contact, and inhalation of fugitive dusts, and surface water and sediment via accidental ingestion and dermal contact. There were no adverse health effects calculated that exceeded USEPA's acceptable value of 1.0 for the adult and adolescent trespassers from exposure to the aforementioned media. These results are presented in Table 7-11.

#### 7.4.3.2 Current/Future On-Site Industrial/Commercial Workers

There were no carcinogenic risks or adverse health hazards calculated that exceeded USEPA's acceptable criteria for the current on-site industrial/commercial worker upon exposure to surface soil (both inside and outside the landfill) via accidental ingestion, dermal contact, and inhalation of fugitive dusts, and surface water and sediment via accidental ingestion and dermal contact. These results are presented in Table 7-12.

#### 7.4.3.3 Future Adult and Younger Child Residents

Table 7-13 presents the carcinogenic and noncarcinogenic risks calculated for potential future on-site adult and child residents (RME scenario) who may be exposed to COPCs in surface soil and subsurface soil (both inside and outside the landfill) via ingestion, dermal contact, and inhalation of fugitive dusts, and groundwater (nonpotable, beneficial use), surface water and sediment via ingestion and dermal contact at Site 1. Table 7-14 presents the carcinogenic and noncarcinogenic risks calculated for potential future on-site adult and child residents under the CT exposure scenarios.

The total ILCR values for Site 1 evaluated for soil inside the landfill under the RME exposure scenario for the adult and young child resident, summed over all media, exceed USEPA acceptable risk range. Under the CT exposure scenario, ILCR value for the child receptor as well as the total lifetime ILCR value exceeded the USEPA acceptable criteria. Total HIs for residential receptors under the RME (adult and child) and CT (child only) exposure scenarios for Site 1 evaluated for soil inside the landfill exceeded 1.0.

The total ILCR values for Site 1 evaluated for soil outside the landfill under the RME exposure scenario for the residential receptors, summed over all media, were within USEPA's acceptable risk range. The total HI for the residential child receptor under the RME exposure scenario for Site 1 evaluated for soil outside the landfill exceeded 1.0. The total HI for the young child resident was also above USEPA's acceptable hazard level of 1.0 when CT exposure scenarios were evaluated.

### *Soil - Inside the Landfill*

The total site ILCRs exceeded USEPA's target risk range of  $1 \times 10^{-04}$  to  $1 \times 10^{-06}$  ( $2.4 \times 10^{-04}$  and  $5.5 \times 10^{-04}$  for the adult and child future residents, respectively) when evaluating the RME exposures using soil inside the landfill. Ingestion of surface soil inside the landfill contributed to these elevated carcinogenic risks. Potential exposures to Site 1 surface soil COPCs inside the landfill via ingestion resulted in ILCR values of  $1.6 \times 10^{-04}$  and  $3.7 \times 10^{-04}$  for the adult and child future residents, respectively, when evaluating the RME exposures. Carcinogenic PAHs via the ingestion exposure pathway were the primary contributors to the exceedences of USEPA's target risk range of  $1 \times 10^{-04}$  to  $1 \times 10^{-06}$ . Benzo(a)pyrene, dibenz(a,h)anthracene, benzo(a)anthracene, and benzo(b)fluoranthene contributed approximately 61 percent, 11 percent, 8 percent, and 8 percent, respectively, to the elevated ILCRs (adult and child) for surface soil inside the landfill. Arsenic contributed 7 percent to the elevated ILCRs for surface soil inside the landfill.

This HHRA also considered exposure to excavated subsurface soil under a future residential scenario. As discussed in Section 7.2.2, it is assumed that subsequent to excavation for residential development, subsurface soil would be brought to the surface. Ingestion of subsurface soil inside the landfill also contributed to the elevated risk. The ILCR for ingestion of subsurface soil inside the landfill for the adult and child were  $7.4 \times 10^{-05}$  and  $1.7 \times 10^{-04}$ , respectively. Carcinogenic PAHs and arsenic contributed primarily to this elevated risk. Benzo(a)pyrene, dibenz(a,h)anthracene, benzo(a)anthracene, and arsenic contributed approximately 55 percent, 11 percent, 6 percent, and 17 percent, respectively, to the elevated subsurface soil ingestion ILCR values for the adult and child. Potential exposure to groundwater, surface water, and sediment did not pose a carcinogenic risk above USEPA's acceptable risk range or contribute significantly to the total site ILCR. These results are presented in Table 7-13.

The total site HIs exceeded USEPA's acceptable level of 1.0 (1.7 and 14.3 for the adult and child future residents, respectively) when evaluating the RME exposures using soil inside the landfill. Ingestion of surface soil and subsurface soil inside the landfill contributed primarily to these elevated HI values. Potential exposures to Site 1 surface soil COPCs inside the landfill via ingestion resulted in HI values of 0.7 and 5.8 for the adult and child future residents, respectively. Antimony, arsenic, copper, and iron concentrations detected in surface soil inside the landfill contributed approximately 25 percent, 14 percent, 10 percent, and 27 percent, respectively, to the elevated surface soil ingestion HIs (adult and child). Potential exposures to Site 1 subsurface soil COPCs inside the landfill via ingestion resulted in HI values of 0.9 and 8 for the adult and child

future residents, respectively. Antimony, arsenic, copper, and iron concentrations were also detected in subsurface soil inside the landfill and contributed approximately 17 percent, 11 percent, 19 percent, and 31 percent, respectively, to the elevated subsurface soil HIs (adult and child).

For the future child resident, the HQ values for antimony (HQ=1.2) and iron (HQ=1.3) in surface soil for the ingestion pathway exceeded 1.0. Also, the HQ values for antimony (HQ=1.1), copper (HQ=1.2), and iron (HQ=2.1) in subsurface soil for the ingestion pathway exceeded 1.0. However, these metals target different organs in the human body. Antimony targets the whole body; arsenic targets the skin; copper targets the gastrointestinal system; and iron targets the liver. Additionally, it is important to note that while iron is evaluated quantitatively in this risk assessment, it is considered an essential nutrient, and the studies that prompted the addition of toxicity criteria are provisional and have not been formally reviewed by the USEPA. Potential exposure to groundwater, surface water, and sediment did not pose a noncarcinogenic hazardous effect above USEPA's acceptable level or contribute significantly to the total site HI. These results are presented in Table 7-13.

#### *Soil - Outside the Landfill*

The total site ILCRs were within USEPA's target risk range of  $1 \times 10^{-4}$  to  $1 \times 10^{-6}$  ( $1.6 \times 10^{-5}$  and  $3.3 \times 10^{-5}$  for the adult and child future residents, respectively) when evaluating the RME exposures using soil outside the landfill. These results are presented in Table 7-13.

The total site HI for the future adult resident (HI=0.47) was less than USEPA's acceptable level of 1.0 when evaluating the RME exposures using soil outside the landfill. However, the total site HI for the future child resident (HI=4.0) exceeded USEPA's acceptable level of 1.0 when evaluating the RME exposures using soil outside the landfill. This elevated HI was primarily due to ingestion pathways for surface soil (HI=1.5) and subsurface soil (HI=1.3) outside the landfill. Arsenic and iron detected in the soil outside the landfill contributed predominantly to the elevated ingestion pathway HI values. It should be noted that only the HQ value for iron exceeded 1.0 when summed over all media. Potential exposure to groundwater, surface water, and sediment did not pose noncarcinogenic adverse health effects above USEPA's acceptable level or contribute significantly to the total site HI. These results are presented in Table 7-13.

#### 7.4.3.4 Future Construction Workers

Table 7-15 presents the carcinogenic risks and noncarcinogenic hazard levels calculated for potential future construction workers who may be exposed to COPCs in surface soil and subsurface soil (both inside and outside the landfill) via ingestion, dermal contact, and inhalation of fugitive dusts, and groundwater via ingestion and dermal contact at Site 1.

##### Soil - Inside the Landfill

The total site ILCR ( $3.4 \times 10^{-5}$ ) was within USEPA's target risk range of  $1 \times 10^{-4}$  to  $1 \times 10^{-6}$  when evaluating the RME exposures using soil inside the landfill. These results are presented in Table 7-15.

The total site HI (4.6) exceeded USEPA's acceptable level of 1.0 when evaluating the RME exposures using soil inside the landfill. Ingestion of surface soil and subsurface soil inside the landfill contributed primarily to these elevated HI values. Potential exposures to Site 1 surface soil COPCs inside the landfill via ingestion resulted in HI value of 1.7. Antimony, arsenic, copper, and iron concentrations detected in surface soil inside the landfill contributed approximately 25 percent, 14 percent, 10 percent, and 27 percent, respectively, to the elevated surface soil ingestion pathway HI. Potential exposures to Site 1 subsurface soil COPCs inside the landfill via ingestion resulted in HI value of 2.5. Antimony, arsenic, copper, and iron concentrations were also detected in subsurface soil inside the landfill and contributed approximately 17 percent, 10 percent, 18 percent, and 31 percent, respectively, to the elevated subsurface soil ingestion pathway HI. It should be noted that the individual HQ values of antimony and iron exceeded 1.0 when summed over all pathways. Antimony targets the whole body, and iron targets the liver. Potential exposure to total groundwater did not pose a noncarcinogenic hazardous effect above USEPA's acceptable level or contribute significantly to the total site HI. These results are presented in Table 7-15.

##### Soil - Outside the Landfill

The total site ILCR ( $2.2 \times 10^{-6}$ ) was within USEPA's target risk range of  $1 \times 10^{-4}$  to  $1 \times 10^{-6}$  when evaluating the RME exposures using soil outside the landfill. These results are presented in Table 7-15.

The total site HI (1.2) was slightly above USEPA's acceptable level of 1.0 when evaluating the RME exposures using soil outside the landfill. It should be noted that none of the exposure pathway HI values exceeded 1.0, also none of the individual HQ values exceeded 1.0. Arsenic, iron, and thallium concentrations detected in surface soil outside the landfill contributed to the surface soil HI of 0.58. Antimony, arsenic, iron, and thallium concentrations detected in subsurface soil outside the landfill contributed to the subsurface soil HI of 0.56. Arsenic, iron, and manganese concentrations detected in total groundwater contributed to the groundwater HI of 0.2. Therefore, it is unlikely that adverse health effects will result from exposure soil outside the landfill for the future construction worker. These results are presented in Table 7-15.

#### 7.4.3.5 Lead IEUBK Model Results

The USEPA lead IEUBK model (Windows™ version) was used to determine if exposure to site media would result in unacceptable blood lead levels in younger children upon exposure to surface soil and subsurface soil inside the landfill at Site 1. Blood lead levels are considered unacceptable if there is a greater than five percent probability that the blood lead levels will exceed 10 µg/dl.

The normal 95 percent UCL concentrations for lead in surface soil (973 mg/kg) and subsurface soil (877 mg/kg) were used in the model. The remaining model parameters used were the default factors supplied in the model (USEPA, 2001). The concentration in surface soil inside the landfill resulted in a 66 percent probability that blood lead levels would exceed 10 µg/dl, which is above the acceptable level. The concentration in subsurface soil inside the landfill resulted in a 42 percent probability that blood lead levels would exceed 10 µg/dl, which is above the acceptable level. These results are presented graphically in Figures 7-2 and 7-3. This indicates that the potential for adverse health effects from exposure to lead may occur in the future child resident.

### 7.5 Sources of Uncertainty

Uncertainties are encountered throughout the process of performing a risk assessment. This section discusses the sources of uncertainty inherent in the following elements of the human health evaluation performed for Site 1:

- Sampling and analysis
- Selection of COPCs

- Exposure assessment
- Toxicity assessment
- Iron
- Risk characterization

In addition, the USEPA stresses the importance of recognizing the unique characteristics and circumstances of each facility and the need to formulate site-specific responses. However, many of the assumptions presented in this document were derived from USEPA guidance, which is designed to provide a conservative approach and cover a broad variety of cases. As such, the generic application of such assumptions to a site in the RME case scenario may work against the objectives of formulating a site-specific response to a constituent presence (i.e., it is possible that the site risks may be over estimated). Uncertainties associated with this risk assessment are discussed in the following paragraphs. Table 7-16 summarizes the potential effects of certain uncertainties on the estimation of human health risks.

#### **7.5.1 Sampling and Analysis**

The development of a risk assessment depends on the reliability of, and uncertainties associated with, the analytical data available to the risk assessor. These, in turn, are dependent on the operating procedures and techniques applied to the collection of environmental samples in the field and their subsequent analyses in the laboratory. To minimize the uncertainties associated with sampling and analysis at Site 1, USEPA approved sampling and analytical methods were employed. Data were generated following USEPA's Statement of Work for Contract Laboratory Program (CLP). Samples were analyzed for Target Compound List (TCL) organics (plus nitramine compounds), Target Analyte List (TAL) inorganics, and cyanide. Samples were taken from locations specified in the approved Work Plan along with the necessary Quality Assurance/Quality Control (QA/QC) samples.

Analytical data are limited by the precision and accuracy of the methods of analysis that are reflected by the Relative Percent Difference (RPD) of duplicate analyses and the percent recovery of spikes, respectively. In addition, the statistical methods used to compile and analyze the data (mean concentrations, detection frequencies) are subject to the overall uncertainty in data measurement. Furthermore, chemical concentrations in environmental media fluctuate over time and with respect to sampling location. Analytical data must be sufficient to consider the temporal and spatial characteristics of contamination at the site with respect to exposure.

Data collected from these sites were validated by an independent third-party validator. Data validation serves to reduce some of the inherent uncertainty associated with the analytical data by establishing the usability of the data to the risk assessor who may or may not choose to include the data point in the estimation of risk. Data qualified as "J" (estimated) were retained for the estimation of risk at these sites. Data can be qualified as estimated for many reasons including a slight exceedence of holding times, high or low surrogate recovery, or intra sample variability. Data qualified "R" (unreliable) were not used in the estimation of risk due to the unusable nature of the data. Due to the comprehensive sampling and analytical program at these sites, the loss of some data points did not significantly increase the uncertainty in the estimation of risk.

#### **7.5.2 Selection of COPCs**

The selection of COPCs is performed in a risk assessment following the evaluation of data. Analytical data also must be comprehensive in order to address the COPCs associated with the site. Types of COPCs encountered at Site 1 include some VOCs, SVOCs, and nitramines. Inorganic constituents were detected in every medium investigated; they were the most dominant class of chemicals detected at Site 1. A summary of the COPC selection criteria is presented below.

- Soil COPCs were selected based on comparisons of the maximum detected concentration with Region III residential soil RBC values.
- Groundwater COPCs were selected based on comparisons of the maximum detected concentration with Region III tap water RBC values.
- Surface water COPCs were selected based on comparisons of the maximum detected concentration to Region III tap water RBC values multiplied by a factor of ten (based on recommendation by Region III toxicologist).
- Sediment COPCs were selected based on comparisons of the maximum detected concentration to Region III residential soil RBC values.

Region III RBC values are based on exposure assumptions and equations that are intended to introduce conservatism in the risk assessment process by changing the COPC screening method

from a relative toxicity screen as presented in RAGS, to an absolute comparison of risk. However, the use of the Region III RBC values, which incorporate a set of non-site-specific assumptions in the selection of COPCs at Site 1, adds conservatism to the baseline HHRA. Furthermore, the use of residential soil RBC values (which are intended for soil not sediment) in the selection of human health COPCs provides a very conservative screening tool.

It should be noted that PAHs, both carcinogenic and noncarcinogenic, were detected in the surface and subsurface soils at Site 1. It is known that PAHs do not occur alone, but rather, as mixtures. In some instances, PAHs can be re-included as COPCs even though they did not exceed screening criteria. Of the carcinogenic PAHs, benzo(a)pyrene is considered the most potent and has the most conservative toxicity criteria (e.g., CSF and RBCs). Furthermore, the toxicity criteria for the other cPAHs is based on the values determined for benzo(a)pyrene through research. Consequently, benzo(a)pyrene was retained as a COPC (if it exceeded screening criteria) to represent a conservative approach. In addition, any other PAH that exceeded criteria was retained.

Currently, no closures are planned for CAX and future residential development is not considered an expected land use for the area. The application of the residential RBC values to soil and groundwater COPC selections would, therefore, tend to result in a list of COPCs that could be considered conservative for a military facility. The use of conservative COPC selections in the baseline HHRA ensures the protection of public health in that the results of the baseline HHRA are incorporated into the determination of remedial alternatives and remedial action objectives in the Feasibility Study (FS).

### **7.5.3 Exposure Assessment**

In performing exposure assessments, uncertainties arise from two main sources. First, uncertainties arise in estimating the fate of a compound in the environment, including estimating release and transport in a particular environmental medium. Second, uncertainties arise in the estimation of chemical intakes resulting from contact by a receptor with a particular medium.

To estimate an intake, certain assumptions must be made about exposure events, exposure durations, and the corresponding assimilation of constituents by the receptor. Exposure factors have been generated by the scientific community and have undergone review by the USEPA. The USEPA has published an Exposure Factors Handbook (USEPA, 1997), which contains the

best and latest values. Regardless of the validity of these exposure factors, they have been derived from a range of values generated by studies of limited numbers of individuals. In all instances, values used in this risk assessment, scientific judgments, and conservative assumptions agree with those of the USEPA.

The use of a RME approach, designed as not to underestimate daily intakes, was employed throughout this risk assessment. The use of 95 percent UCL estimates of the arithmetic mean versus maximum values as the concentration term in estimating the CDI or DAD for soil, surface water and sediment exposure scenarios reduces the potential for underestimating exposure at Site 1. The use of the maximum concentration as the exposure point concentration for the groundwater data in estimating the DAD or CDI for future groundwater usage was selected based upon the distribution of the highest concentration of contamination. Although the use of the maximum concentration may result in an overestimation of potential risks associated with the site, the exposure point concentrations selected causes the estimation of CDIs and DADs to err on the side of health conservatism.

The USEPA Region III has adopted a non-evaluation approach for PAH exposures via the dermal pathway. Studies have shown that effects due to absorption of PAHs may not be significant since adverse effects are typically observed at the point of application. It is not expected that the exclusion of the dermal pathway from the evaluation of PAH exposure will underestimate potential risk to human health.

#### **7.5.4 Toxicological Assessment{tc \13 "6.5.4 Toxicological Assessment}**

In making quantitative estimates of the toxicity of varying dosages of compounds to human receptors, uncertainties arise from two sources. First, data on human exposure and the subsequent effects are usually insufficient, if they are at all available. Human exposure data usually lack adequate concentration estimations and suffer from inherent temporal variability. Therefore, animal studies are often used and new uncertainties arise from the process of extrapolating animal results to humans. Second, to obtain observable effects with a manageable number of experimental subjects, high doses of a compound are often used. In this situation, a high dose means that high exposures are used in the experiment with respect to most environmental exposures. Therefore, when applying the results of the animal experiment to the human condition, the effects at the high doses must be extrapolated to approximate effects at lower doses. In extrapolating effects from high doses in animals to low doses in humans,

scientific judgment and conservative assumptions are employed. In selecting animal studies for use in dose-response calculations, the following factors are considered:

- Studies are preferred where the animal closely mimics human pharmacokinetics.
- Studies are preferred where dose intake most closely mimics the intake route and duration for humans.
- Studies are preferred which demonstrate the most sensitive response to the compound in question.

For compounds believed to cause threshold effects (i.e., noncarcinogens) safety factors are employed in the extrapolation of effects from animals to humans and from high doses to low doses. In deriving carcinogenic potency factors, the 95 percent UCL value is promulgated by the USEPA to prevent underestimation of potential risk.

Further conservatism in the baseline HHRA is also introduced through the use of experimentally-derived oral absorption efficiencies to account for a difference in the degree of toxicity between an administered dose and an absorbed dose. Equating the absorption efficiency of the dermal bi-phasic barrier to the absorption efficiency of the gastrointestinal lining is a very conservative approach that tends to overestimate the potential risk to human health.

In summary, the use of conservative assumptions, results in quantitative indices of toxicity that are not expected to underestimate potential toxic effects, but may overestimate these effects by an order of magnitude or more.

### **7.5.5 Iron**

The element iron has been given a PRG value and toxicity values with which to evaluate potential human health risks. The studies that prompted the addition of a RBC value for iron are provisional only and have not undergone formal review by the USEPA. A provisional RfD has been derived for iron by the Superfund Technical Support Center (STSC) division of the Environmental Criteria and Affects Office. The provisional RfD is based on a “no observed adverse effect level” (NOAEL). Developing an RfD for iron is problematic because the dose-response curve for iron is “U-shaped”. That is, health effects such as anemia occur at low doses

due to deficiency (occurring in the U.S. in approximately 3.3 million women of childbearing age and 240,000 children aged 1-2 years), and high doses can produce toxic effects such as hemosiderosis and liver cirrhosis, while doses in between are beneficial for most of the population. The NOAEL is based on a study that compared the average intake of iron in the American population with biochemical indices of iron in blood (Looker et al., 1988) to demonstrate that the average intake was sufficient to prevent iron deficiency and insufficient to cause toxic effects of iron overload. The NOAEL (0.15 to 0.27 mg/kg-day) is divided by an uncertainty factor of 1, since iron is an essential element, to produce a provisional RfD of 0.3 mg/kg-day (STSC, 1999).

Although the STSC (1999) report places a high confidence in the critical study upon which the RfD is based; they place a medium confidence in the RfD. The RfD is reported to supply adequate levels of iron to meet the lifetime nutritional requirements for adults and adolescents but may not be protective of people with inherited disorders of iron metabolism (e.g., hemochromatosis which occurs in up to one million individuals in the U.S.) (MMWR, 1998) and could be conservative if applied to exposure scenarios involving forms of iron with low bioavailability. This last point is borne out by studies of Ethiopian populations that have the highest per capita iron intake in the world (471 mg/day average daily intake) but for which adverse health effects have not been observed. This is attributed to the low bioavailability of the iron in Ethiopian food (STSC, 1999).

As applied to an incidental soil ingestion exposure scenario, it is important to note that the contribution of intake of iron from soil is expected to be minimal compared to dietary intake. For example, assuming soil with iron concentration of 15,000 mg/kg (a conservative estimate of background concentrations of iron in soil) and ingestion of 50 mg/day for adults, produces only 0.01 mg/kg-day iron from soil compared to a normal dietary level of 0.3 mg/kg-day. Furthermore, the bioavailability of iron from minerals in soil is expected to be significantly lower than the bioavailability of iron from food. (However, actual levels of bioavailability of iron from soil are not known.) For these reasons, and the fact that the primary sensitive population is those individuals with the medical condition of hemochromatosis which is caused by abnormal absorption of iron and which appears to occur irrespective of excess iron intake, the iron RfD is considered very conservative for use in risk assessment from environmental exposures and should be interpreted with considerable uncertainty.

For the scenarios where the HI values exceeded 1, iron had relatively large HQ values. If the provisional iron RBC value were reduced, a large proportion of the risk for these sites would be eliminated. However, by evaluating iron in the risk assessment, a conservative approach is taken and potential toxic effects may be estimated.

In summary, the use of conservative assumptions results in quantitative indices of toxicity that are not expected to underestimate potential toxic effects, but may overestimate these effects by an order of magnitude or more.

#### **7.5.6 Human Risk Characterization**

The risk characterization bridges the gap between potential exposure and the possibility of systemic or carcinogenic human health effects, ultimately providing impetus for the remediation of the site or providing a basis for no remedial action.

Uncertainties associated with risk characterization include the assumption of chemical additivity and the inability to predict synergistic or antagonistic interactions between COPCs. These uncertainties are inherent in any inferential risk assessment. USEPA recommended inputs to the quantitative risk assessment and toxicological indices are calculated to be protective of the human receptor and to err conservatively in an effort not to underestimate the potential human health risks.

#### **7.6 Summary and Conclusions**

##### Current Receptors

There were no carcinogenic risks or adverse health hazards calculated that exceeded USEPA's acceptable criteria for the current on-site adult and adolescent trespassers and on-site industrial/commercial worker upon exposure to surface soil (both inside and outside the landfill) via accidental ingestion, dermal contact, and inhalation of fugitive dusts, and surface water and sediment via accidental ingestion and dermal contact. It is unlikely that adverse health effects would occur for these receptors upon exposure to Site 1 media.

### Future Receptors

Future on-site adult and child residents and future construction workers were evaluated for potential exposure to surface soil and subsurface soil inside and outside the landfill via ingestion, dermal contact, and inhalation of fugitive dusts, and groundwater (nonpotable, beneficial use) at Site 1. Future residents were also evaluated for potential exposure to surface water and sediment via ingestion and dermal contact at Site 1.

The total residential ILCR values for inside the landfill exceeded USEPA acceptable criteria. The elevated ILCRs were due to concentrations of benzo(a)pyrene, dibenz(a,h)anthracene, benzo(a)anthracene, benzo(b)fluoranthene, and arsenic in soil inside the landfill. Total residential HIs for inside the landfill exceeded 1.0. The elevated HIs were due to concentrations of antimony, arsenic, copper, and iron in soil inside the landfill. Also the results of the IEUBK model used to evaluate the concentrations of lead in soil inside the landfill indicated that adverse health effects may occur subsequent to exposure to soil inside the landfill at Site 1.

The total residential ILCR values for outside the landfill were within USEPA's acceptable risk range. The total HI for the residential child receptor exceeded 1.0. Arsenic and iron detected in the surface soil and subsurface soil outside the landfill contributed to the elevated HI.

The total site ILCR values the future construction worker receptor, summed over all media, were within USEPA's acceptable risk range. The total site HI for the construction worker receptor inside the landfill exceeded 1.0. The elevated HIs were due to concentrations of antimony, arsenic, copper, and iron in soil inside the landfill. The total HI for the construction worker receptor outside the landfill exceeded 1.0. Arsenic and iron detected in the soil outside the landfill contributed to the elevated HI.

Based on the ILCR and HI values that exceed USEPA's acceptable risk range and hazard level of 1.0 for the exposure scenarios that include soil inside the landfill for the future receptors and the high concentrations of PAHs and certain inorganics, adverse health effects could occur to the future human receptors upon exposure to the landfill material. The elevated carcinogenic risks and hazard indices were primarily driven by the soil ingestion pathway. It should be noted that it is unlikely that this site will be developed for future residential use.

## 8.0 ECOLOGICAL RISK ASSESSMENT

This section presents a Screening-Level Ecological Risk Assessment (SERA) and Step 3a addressing potential risks to ecological receptors at Site 1, Cheatham Annex Site (CAX), Williamsburg, Virginia. Ecological risk assessment, as defined by the United States Environmental Protection Agency's (USEPA's) *Framework for Ecological Risk Assessment*, is a process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors (USEPA, 1992d). The potential stressors identified in association with Site 1 are chemicals that have been released to environmental media on the site or in nearby areas. This assessment was conducted in accordance with the Navy Tier II Ecological Risk Assessment (ERA) approach developed with EPA Region III, which is based on the USEPA Ecological Risk Assessment Guidance (USEPA, 1997b) and the Chief of Naval Operations (CNO) Policy (CNO 1999). The Navy ERA process consists of eight steps organized into three tiers and is a clarification and interpretation of the eight-step ERA process outlined in the USEPA ERA guidance for the Superfund program (USEPA, 1997b). Under Navy policy, if the results of Step 1 and Step 2 (Tier 1) indicate that there are chemicals present in environmental media that may present a risk to receptor populations and communities based on a set of conservative exposure assumptions, then the ERA process proceeds to the baseline ERA (Tier 2). In the first step of Tier 2 (Step 3a), assumptions applied in Tier I are refined and risk estimates are recalculated using the same conceptual site model. These actions precede the baseline risk assessment problem formulation. The evaluation of risks in Step 3a may also include consideration of background data, chemical bioavailability, and the frequency at which chemicals were detected.

The SERA portion of this report constitutes the first two steps of the eight-step process (Tier 1 and the first step of Tier 2 of the Navy ERA process). Step 1 is Screening-Level Problem Formulation and Ecological Effects Evaluation, and Step 2 is Screening-Level Exposure Estimate and Risk Calculation. This report also presents Step 3a, a Refinement of Conservative Exposure Assumptions. In Step 3a, exposure assumptions are refined and risk estimates are recalculated to provide a more realistic, less conservative evaluation of site risks. The conclusions of this assessment will be used to determine if future investigations or remedial actions at Site 1 are warranted for the overall protection of the environment.

The objectives of the SERA were as follows:

- Screen groundwater, soil, sediments, and surface water to determine if potential risks to ecological receptors warrant either
- additional investigation beyond the conservative screening steps and refinement of the ERA process (unacceptable ecological risks possible) or
- the removal of the site from further ecological consideration (no unacceptable risks).
- Identify any gaps or areas of unacceptable uncertainty that might require the collection of additional data to support ERA evaluations beyond Step 3a.

## **8.1 Screening-Level Problem Formulation**

Problem formulation establishes the goals, scope, and focus of the SERA. As part of problem formulation, the environmental setting of a site is characterized in terms of the habitats and biota known or likely to be present, including the presence of rare, threatened, or endangered species or habitats. Types and concentrations of chemicals that are present in ecologically relevant media are also characterized. A conceptual model is developed for the site that describes potential sources, exposure pathways, exposure routes, and receptors. Assessment endpoints, measurement endpoints, and risk hypotheses are then selected to evaluate those receptors that could be adversely affected by contaminants from the site. The fate, transport, and toxicological properties of the chemicals present at a site are also considered during this process.

### **8.1.1 Environmental Setting**

Site 1 is located at the junction of the York River and a small, unnamed tributary to the York River (See Figure 2-2). The creek originates at a culvert draining the warehouse area of CAX and meanders through a freshwater, forested wetland before approaching the western border of Site 1. This portion of the stream provides potentially significant foraging habitat and a drinking water source for mammalian and avian consumers. This tributary is surrounded by fresh and then saltwater emergent wetlands as it flows towards the York River and is separated from the main landfill by the large debris pile just upstream of the York River. The creek supports wetland vegetation, aquatic invertebrates, amphibians, and small fish in each of its sections. The lower reaches of the creek are under tidal influence from the York River. Vegetation in this area is typical of salt marshes along the east coast. Section 3.7.2.2 describes the aquatic environment of Site 1 in greater detail.

The terrestrial environment of Site 1 can be delineated into two distinct habitats. The main landfill area is flat, indicative of the cap that was installed in 1981, and is covered with various grasses. This area steeply descends to the York River to the north and to a dense mixed hardwood forest to the south. Exposed landfill material can be found all along the southern and western slope. Section 3.7.2.1 describes the terrestrial environment of Site 1 in greater detail.

### **8.1.2 Site History and Available Analytical Data**

Section 2.2.1.2 describes the history of Site 1 in detail. Two previous ecological risk screens have been performed for Site 1. The Site Screening Process Report (Baker, 1997) and the Field Investigation Report (Baker, 1999) both included media screens of known contaminants from a limited data set for lower trophic level receptors. Neither investigation included an assessment of upper trophic level receptors or Step 3a.

Data used in the SERA comes from the 1998 and 2000 field investigation associated with the Draft Remedial Investigation, from additional sampling conducted in the Summer of 2001 to better delineate the nature and extent of contamination in the unnamed tributary bordering the site, and from the Fall 2001 Trenching Study performed in the main landfill and surrounding terrestrial environment. Table 8-1 outlines the data evaluated in the ERA. A map showing the spatial relationship of the sampling locations to the site and to the bordering estuarine wetland system is provided as Figure 8-1.

In order for data to be considered appropriate for use in the SERA, a qualified data validator using acceptable data validation methods must have validated the data. Data with rejected (R) values were not used in the risk assessment. Unqualified data and data qualified as K, L, J or N were treated as detected. Data qualified as B, U, or UJ were treated as non-detected.

Site-specific considerations also guided the data selection process, and are noted in the following sections. Gaps in the data required to perform a SERA for Site 1 are also noted below. The data available for the Site 1 SERA are summarized in Table 8-2 and include 1998 surface water, sediment data, 2000 groundwater data, and 2001 soil, surface water, and sediment data. Analytical data used in the SERA are provided and discussed in Section 5.5.

Groundwater. Groundwater data collected in 2000 were used to evaluate the potential for contaminants to migrate to surface water bodies at concentrations sufficient to pose unacceptable aquatic risks. A discussion of laboratory analytical results and nature and extent of constituent contamination is presented in Section 5.5 of this report. Only phthalate esters, the explosive 4-nitrotoluene and inorganics were detected in Site 1 groundwater. Groundwater sample 1-GW05, located upgradient of the site, was used for comparison of on-site background concentrations in Step 3a.

Soils. The main portion of the landfill surface was regraded and covered with two feet of soil in 1981. Samples were collected in 2001 from the fringe areas surrounding this cap and from the drainage swale bordering the southern edge of the landfill (see Figure 8-1). Soil samples were divided into surface samples (0 to 6 inches [in.] below ground surface [bgs]) and subsurface samples (6 to 24 in. bgs). Analytical data from these areas of the landfill suggests that chemicals may be present in ecologically relevant concentrations in the surface soils at the site. Volatile organic compounds (VOCs), polycyclic aromatic hydrocarbons (PAHs) and other semivolatile organic compounds (SVOCs), pesticides, polychlorinated biphenyls (PCBs), and inorganics were all detected. To address the potential, surface soil data collected in the Fall 2001 trenching investigation were combined with the samples taken from the drainage swale bordering the southern edge of the landfill to characterize and evaluate Site 1 surface soils as a whole. Subsurface soils samples were only collected from the swale area. SVOCs, pesticides, PCBs, and inorganics were detected in both surface and subsurface samples collected from the drainage swale. The VOC 1,1,2,2-tetrachloroethane was also detected in one surface soil drainage swale sample. Samples collected during the 2001 Base-wide Background Investigation (Baker, 2002) from similar soils at CAX were used for the comparison of site contaminant concentrations to background levels in Step 3a.

Surface water and sediments. Both the 1998 and 2001 sediment data from the stream and marsh bordering the site were used to characterize exposures to aquatic receptors. In addition to surface water and sediment samples located with the stream in 2001, seven sediment samples were preferentially collected from seepage and depositional areas in the marsh at the foot of the landfill's western edge. Sediment samples were separated into surface samples (0 to 4 in. bgs) and subsurface samples (4 to 8 in. bgs). Surface water and sediment samples were grouped and analyzed according to their location in relation to the landfill and according to habitat type (see Table 8-1). Samples located upstream of the saltwater influence of the creek were screened against freshwater surface water and sediment screening values. Those located within the

estuarine influence of the creek were screened against saltwater surface water and sediment screening values. Wetland vegetation and salinity measurements taken in the summer of 2001 were used to delineate the freshwater (average low tide salinities <1.0 parts per thousand [ppt]) and saltwater habitats (salinities >15.0 ppt) at the site. Samples collected from beyond and upgradient of the Site's influence (samples 1-SW/SD01 and CX01SW/SD01 on Figure 8-1) were evaluated as on-site background samples in Step 3a.

Only inorganics and methyl acetate (one location) were detected in Site 1 surface waters. VOCs, SVOCs, pesticides, PCBs, and inorganics were found in 1998 and 2001 sediments.

#### 8.1.2.1 Fate and Transport Mechanisms of Potential Contaminants of Concern

Measured groundwater, surface water, sediment, and surface soil concentrations reflect the acting fate and transport mechanisms of the ecological Contaminants of Potential Concern (COPCs) at the site and provide a direct means to characterize exposure to the abiotic media. In the absence of measured values (e.g., for biotic media), the transport and partitioning of constituents into particular environmental compartments, and their ultimate fate in those compartments, can be predicted from key physio-chemical characteristics. The physio-chemical characteristics that are most relevant for exposure modeling in this assessment include water solubility, adsorption to solids, octanol-water partitioning, and degradability. These characteristics are defined below.

The water solubility of a compound influences its partitioning to aqueous media. Highly water-soluble constituents, such as most volatile organics, have a tendency to remain dissolved in the water column rather than partitioning to soil or sediment (Howard, 1991). Compounds with high water solubility (e.g., volatile organic compounds [VOCs]) also generally exhibit a lower tendency to bioconcentrate in aquatic organisms and a greater likelihood of biodegradation, at least over the short term (Howard, 1991).

Adsorption is a measure of a compound's affinity for binding to solids, such as soil or sediment particles. Adsorption is expressed in terms of partitioning, either  $K_d$  (adsorption coefficient; a unitless expression of the equilibrium concentration in the solid phase versus the water phase) or as  $K_{oc}$  ( $K_d$  normalized to the organic carbon content of the solid phase; again unitless) (Howard, 1991). For a given organic chemical, the higher the  $K_{oc}$  or  $K_d$ , the greater the tendency for that chemical to adhere strongly to soil or sediment particles.  $K_{oc}$  values can be measured directly or

can be estimated from either water solubility or the octanol-water partition coefficient using one of several available regression equations (Howard, 1991).

Octanol-water partitioning indicates whether a compound is hydrophilic or hydrophobic. The octanol-water partitioning coefficient ( $K_{ow}$ ) expresses the relative partitioning of a compound between octanol (lipids) and water. A high affinity for lipids equates to a high  $K_{ow}$  and vice versa.  $K_{ow}$  has been shown to correlate well with Bioconcentration Factors (BCFs) in aquatic organisms, adsorption to soil or sediment particles, and the potential to bioaccumulate in the food chain (Howard, 1991). Typically expressed as  $\log K_{ow}$ , a value of three (3.0) or less generally indicates that the chemicals will not bioconcentrate to a significant degree (Maki and Duthie 1978). A  $\log K_{ow}$  of three equates to an aquatic species BCF of about 100, using the equation (Lyman et al. 1990):

$$\log BCF = (0.76) (\log K_{ow}) - 0.23$$

Degradability is an important factor in determining whether there will be significant loss of mass or change in the form of a constituent over time in the environment. The half-life of a compound is typically used to describe losses from either degradation (biological or abiotic) or from transfer from one compartment to another (e.g., volatilization from soil to air). The half-life is the time required for one-half of the mass of a compound to undergo the loss or degradation process.

### 8.1.3 Preliminary Conceptual Model

The conceptual site model is used to identify complete exposure pathways. Exposure, and thus potential for risk, can only occur if each of the following conditions exist (USEPA, 1998c):

- A source of contamination must be present.
- Release and transport mechanisms must be available to move the contaminants from the source to an exposure point.
- An exposure point must exist where ecological receptors could contact the affected media.
- An exposure route must exist whereby the contaminant can be taken up by ecological receptors.

A preliminary conceptual model for Site 1 is provided as Figure 8-2. Potentially complete and significant transport pathways are shown as solid lines if they were evaluated quantitatively in this investigation. Dashed lines represent pathways that were incomplete, complete but insignificant, or were not evaluated in this investigation. Potentially complete and significant exposure pathways were not evaluated if a potential receptor group (e.g., amphibians) was not selected as an assessment endpoint.

#### 8.1.3.1 Source of Contamination

Chemicals originating from the Landfill are the source of contamination at this site. Though the main landfill is covered with two feet of soil, exposed debris on the slopes and a large debris pile on the western edge bordering the unnamed tributary remain. Each of these sources has the potential to release contaminants into the surrounding surface and subsurface soils.

#### 8.1.3.2 Exposure Pathways

An exposure pathway describes the course a contaminant takes from the source to an exposed organism. Each exposure pathway contains a source, an exposure point, and an exposure route. If the exposure point differs from the source, transport/exposure media are also included (USEPA, 1997b). Exposure, and thus potential risk, can only occur if complete exposure pathways exist.

As stated above, the source of contamination at Site 1 is the debris that was disposed at the site and any releases of contaminants from that debris. Contamination from the source is likely to have impacted surface and perhaps subsurface soils. These soils are one of the exposure points at the site. Contamination may also have been transported to other media in the site area. Precipitation may have resulted in leaching of contaminants to groundwater flowing below the site. Surface waters and sediments may have been contaminated by the discharge of contaminated groundwater and/or by run-off of surface soils to aquatic habitats with overland surface water flow following precipitation events. Run-off may have also transported contaminated soils to down-gradient terrestrial areas. Site-related chemicals in soil, sediment, and surface water may be taken up and accumulated in the tissue of biota, and thus transported to upper trophic level receptors via food webs. Thus, exposure points at Site 1 include surface soils, surface water, sediment, and food items.

Groundwater does not represent an exposure point because receptors are not exposed directly to groundwater. However, groundwater may transport contamination to aquatic habitats. It should be noted that any current groundwater impact on the aquatic habitat will be reflected in surface water and sediment samples collected at the site. As a measure of conservatism, groundwater data are also screened for ecological effects assuming discharge to a surface water body with no dilution or natural attenuation.

An exposure route describes the specific mechanism(s) by which a receptor is exposed to a chemical present in an environmental medium. Terrestrial plants may be exposed through their root surfaces during water and nutrient uptake to chemicals present in surface soils. Unrooted, floating aquatic plants, and rooted submerged vascular aquatic plants and algae, may be exposed to chemicals directly from the water or (for rooted plants) from sediments. Animals may be exposed to chemicals through: (1) the inhalation of gaseous chemicals or of chemicals adhered to particulate matter; (2) the incidental ingestion of contaminated abiotic media (e.g., soil or sediment) during feeding activities; (3) the ingestion of contaminated water; (4) the ingestion of contaminated plant and/or animal tissues for chemicals which have entered food webs; and/or (5) dermal contact with contaminated abiotic media. These exposure routes, where applicable, are depicted on Figure 8-2.

The exposure routes evaluated in this ERA for upper trophic level receptors are as follows.

- Ingestion of contaminated plant and/or animal tissues.
- Incidental ingestion of surface soil and sediment.
- Ingestion of surface water.

Direct ingestion of drinking water is only considered at an ecological area of concern if upper trophic level receptors were selected for evaluation and if the area of concern has on-site or contiguous drinking water source. Surface water analytical data from the freshwater portion of the unnamed tributary bordering the west side of Site 1 was used to evaluate drinking water exposures for upper trophic level ecological receptors.

Based on the chemicals present at Site 1 (generally metals, pesticides, PAHs, and PCBs), their general fate properties (e.g., relatively high adsorption to solids), and the protection offered by hair (mammals) and feathers (birds), dermal exposures for upper trophic level receptor species are not considered significant relative to ingestion exposures. Therefore, dermal exposures are not

evaluated in the SERA. Direct contact is considered for lower trophic level receptors (e.g., fish, terrestrial invertebrates, and benthic invertebrates), however. Similarly, based on the limited burrowing habitat available and the lack of exposed soil at the site, inhalation exposures were assumed to be insignificant when compared to ingestion exposures and were not evaluated in the Tier 1 evaluation.

#### 8.1.3.3 Endpoints and Risk Hypotheses

The conclusion of the screening-level problem formulation includes the selection of ecological endpoints, which are based on the conceptual model. Endpoints in the SERA define ecological attributes that are to be protected (assessment endpoints) and a measurable characteristic of those attributes (measurement endpoints) that can be used to gauge the degree of impact that has or may occur. Assessment endpoints most often relate to attributes of biological populations or communities, and are intended to focus the risk assessment on particular components of the ecosystem that could be adversely affected by chemicals attributable to the site (USEPA, 1997b). Assessment endpoints contain an entity (e.g., muskrat population) and an attribute of that entity (e.g., survival rate). Individual assessment endpoints usually encompass a group of species or populations (the receptor) with some common characteristic, such as specific exposure route or contaminant sensitivity, with the receptor then used to represent the assessment endpoint in the risk evaluation. The considerations for selecting assessment and measurement endpoints are summarized in USEPA (1992d, 1997b) and discussed in detail in Suter (1989, 1990, and 1993). Risk hypotheses are testable hypotheses about the relationship among the assessment endpoints and their predicted responses when exposed to contaminants.

Assessment and measurement endpoints may involve ecological components from any level of biological organization, from individual organisms to the ecosystem itself (USEPA, 1992d). Effects on individuals are important for some receptors, such as rare and endangered species. Population- and community-level effects are typically more relevant to ecosystems. Population- and community-level effects are usually difficult to evaluate directly without long-term and extensive study. However, measurement endpoint evaluations at the individual level, such as an evaluation of the effects of chemical exposure on reproduction, can be used to predict effects on an assessment endpoint at the population or community level. In addition, use of criteria values designed to protect the vast majority (e.g., 95 percent) of the components of a community (e.g., National Ambient Water Quality Criteria for the Protection of Aquatic Life [NAWQC]) can be useful in evaluating potential community- and/or population-level effects.

Preliminary assessment endpoints, risk hypotheses, and measurement endpoints selected for the SERA are presented in Table 8-3. The assessment endpoints selected were the survival, growth, and reproduction of lower trophic level receptor groups (terrestrial plants and invertebrates, aquatic plants, benthic invertebrates, fish, and amphibians) and upper trophic level mammals and birds (herbivores, omnivores, and carnivores). The population traits of interest for each of the assessment endpoints (survival, growth, and reproduction) represent components of a healthy population. Failure or impairment of survival, growth, or reproduction will adversely affect the ability of the population to be healthy and viable and fill its appropriate role in an ecosystem.

Measurement endpoint for terrestrial/aquatic plants, soil/benthic invertebrates, fish, and amphibians is the comparison of maximum soil, surface water, and/or sediment concentrations to medium-specific screening values. The measurement endpoint for the upper trophic level receptors is the comparison of the literature-derived chronic NOAEL values for survival, growth, and reproductive effects with modeled dietary exposure based on maximum soil, surface water, and/or sediment concentration.

Because of the complexity of natural systems, it is generally not possible to directly assess the potential impacts to all ecological receptors present within an area. Therefore, specific receptor species (e.g., great blue heron) or species groups (e.g., fish) are often selected as surrogates to evaluate potential risks to larger components of the ecological community (e.g., piscivorous birds) that were used to represent the assessment endpoints (e.g., survival, growth, and reproduction of piscivorous birds). Receptor species selected for evaluation typically include those species that:

- Are known to occur, or are likely to occur, at the site
- Have a particular ecological, economic, or aesthetic value
- Are representative of taxonomic groups, life history traits, and/or trophic levels in the habitats present at the site for which complete exposure pathways are likely to exist
- Can, because of toxicological sensitivity or potential exposure magnitude, be expected to represent potentially sensitive populations at the site
- Have sufficient ecotoxicological information available on which to base an evaluation

Since potentially complete exposure pathways exist for reptiles at Site 1 (see Figure 8-2), reptiles were selected as receptors despite a paucity of data concerning the toxicological effects of

chemicals on this group via ingestion exposures. Reptilian receptors could not be quantitatively evaluated because data regarding exposure parameters and toxicity data for these receptors are limited. However, it can be qualitatively stated that reptiles are not at risk if no risks are identified in either the potential food sources or predators of reptiles. Although this represents an uncertainty in the assessment, it is assumed that this group of organisms is not likely to be more sensitive to chemical exposures than other receptor groups that are included in the SERA.

Lower trophic level receptor species are evaluated in the SERA based on those taxonomic groupings for which screening values have been developed; these groupings and screening values are used in most ecological risk assessments. These groupings and screening values are used in most ERAs. As such, specific species of aquatic biota (e.g., fish and macroinvertebrates) are not chosen as receptor species because of the limited information available for specific species and because aquatic biota are dealt with on a community level via a comparison to surface water and sediment screening values. Similarly, terrestrial plants and soil invertebrates (earthworms are the standard surrogate) are evaluated using soil screening values developed specifically for these groups.

The upper trophic level receptor species listed below have been chosen for dietary exposure modeling based on the criteria listed above, the general guidelines presented in USEPA (1991c), and the assessment endpoints (see Table 8-3). Only those receptors that were likely present or for which the necessary habitat is present were included at Site 1. For example, both the eastern screech owl and the red-tailed hawk are avian carnivores that are likely to occur at Site 1, as both wooded and more open foraging habitats are available. The eastern screen owl was selected to represent avian carnivores and will be evaluated at Site 1, however, as the dominant habitat type is more wooded in nature and the owl is likely to forage in the open fringe areas as well. Similarly, though Site 1 contains bluff habitat that may be suitable for the belted kingfisher, the great blue heron was chosen as an aquatic avian piscivore as the dominant aquatic habitat type within the boundaries of the site is emergent marsh where wading herons are more likely to occur.

*Terrestrial species:*

- American robin (*Turdus migratorius*) - avian omnivore
- Eastern screech owl (*Otus asio*) - avian carnivore, wooded areas
- Meadow vole (*Microtus pennsylvanicus pennsylvanicus*) - mammalian herbivore

- Mourning dove (*Zenaida macroura*) - avian herbivore
- Red fox (*Vulpes vulpes*) - mammalian carnivore
- Short-tailed shrew (*Blarina brevicauda*) - mammalian insectivore
- White footed mouse (*Peromyscus leucopus*) - mammalian omnivore

*Aquatic/Wetland species:*

- Great Blue Heron (*Ardea herodias herodias*) - semiaquatic avian opportunistic piscivore
- Mallard (*Anas platyrhynchos*) - aquatic avian omnivore
- Marsh wren (*Cistothorus palustris*) - avian insectivore
- Mink (*Mustela vison*) - semiaquatic mammalian piscivore
- Muskrat (*Ondatra zibethicus*) - semiaquatic mammalian herbivore
- Raccoon (*Procyon lotor*) - semiaquatic mammalian omnivore

## **8.2 Screening-Level Effects Evaluation**

The purpose of the screening-level effects evaluation is to establish chemical exposure levels (screening values) that represent conservative thresholds for adverse ecological effects. One set of screening values is typically developed for each selected assessment endpoint. For this evaluation, two types of screening values were developed. Media-specific screening values were developed for groundwater, surface soil, surface water, and sediment. Ingestion-based screening values were developed for upper trophic level receptors.

### **8.2.1 Media-Specific Screening Values**

Medium-specific screening values are established for ecologically relevant media including groundwater, surface soil, surface water (fresh and saltwater), and sediment (fresh and saltwater).

The groundwater, surface soil, surface water, and sediment screening values used in the SERA are USEPA Region III Biological Technical Assistance Group (BTAG) screening values (USEPA 1995a) (see Table 8-4). Where more than one final screening value was available for a specific medium and chemical (e.g., soil flora and fauna), the lowest of these values was conservatively selected for use in the SERA. As noted in the table, BTAG values have been updated to reflect current data. For example, surface water screening values based on USEPA NAWQC have been

updated to reflect the most recent criteria. If no BTAG value was available for a given chemical, secondary literature sources were consulted to obtain a screening value.

Ecological receptors are not directly exposed to groundwater; however, they may be impacted by groundwater contamination that discharges to a surface water body. Because surface water and sediment data were available at Site 1, the potential migration of chemicals with groundwater could be directly evaluated through evaluation of the surface water and sediment data. In addition, as a conservative measure, groundwater data were screened against freshwater surface water screening values. This screening provides information regarding the potential for groundwater to adversely impact the aquatic habitat. Freshwater surface water values were in the groundwater comparison because the Field Investigation Report for Site 1 (Baker 1999) indicated that only monitoring well 1-GW06 fluctuated with tidal elevations (approximately 1 inch). No other wells are under tidal influence. In addition, the surface water body first encountered by groundwater discharge would be freshwater sections of the unnamed tributary.

A number of the freshwater screening values for certain metals (cadmium, copper, lead, nickel, and zinc) within Table 8-4 are based on the hardness of the surface water. Hardness-dependent surface water screening values were derived using an equation from Franson (1992) that calculates hardness-based criteria using total calcium and magnesium concentrations in surface water:

$$\text{Hardness (as CaCO}_3\text{)} = 2.497 (\text{Ca}) + 4.118 (\text{Mg})$$

Where: CaCO<sub>3</sub> = Calcium carbonate  
Ca = Calcium concentration (mg/L)  
Mg = Magnesium concentration (mg/L)

Site 1 hardness-dependent surface freshwater screening values were adjusted using the minimum of the three values measured in this habitat (335 mg/L, 337 mg/L, and 395 mg/L). The hardness value was used to calculate surface water screening values (SWSV) for total recoverable metals as follows (USEPA, 1999b):

- Cadmium SWSV = EXP(0.7852\*(LN(hardness))-2.715)
- Copper SWSV = EXP(0.8545\*(LN(hardness))-1.702)
- Lead SWSV = EXP(1.273\*(LN(hardness))-4.705)

- Nickel SWSV =  $\text{EXP}(0.846 * (\text{LN}(\text{hardness})) + 0.0584)$
- Zinc SWSV =  $\text{EXP}(0.8473 * (\text{LN}(\text{hardness})) + 0.884)$

In the SERA, only total recoverable metals data for surface water and groundwater were considered. Conversion factors provided by the USEPA (1999b) were used to adjust screening values for dissolved metals to values that are appropriate for comparison to data on total metals fractions. The use of total metals data in the SERA is a conservative measure. For some metals (arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, and zinc), only the dissolved fraction is bioavailable; therefore, the use of total recoverable metals data for these compounds is likely to overestimate potential risks. In the case of groundwater, the dissolved fraction of all metals is more likely to represent the concentration of metals that may migrate with groundwater flow and discharge to a surface water body. Dissolved metals data are considered in Step 3a.

Although calcium, magnesium, potassium, and sodium were analyzed in the samples collected from Site 1, these inorganics were not evaluated in the SERA. As such, screening values from these chemicals are not shown in Table 8-4. They have been excluded from evaluation because they are essential nutrients (Robbins 1983) with very low toxicity (USEPA 1989).

### **8.2.2 Ingestion-Based Screening Values**

Ingestion-based screening values for dietary exposures were derived for each avian/mammalian receptor species and bioaccumulating chemical. Toxicological information from the literature for wildlife species most closely related to the receptor species was used, where available, but was supplemented by laboratory studies of non-wildlife species (e.g., laboratory mice) where necessary. The ingestion-based screening values are expressed as milligrams of the chemical per kilogram body weight of the receptor per day (mg/kg-BW/day).

Growth and reproduction were emphasized as assessment endpoints since they are the most relevant, ecologically, to maintaining viable populations and because they are generally the most studied chronic toxicological endpoints for ecological receptors. If several chronic toxicity studies were available from the literature, the most appropriate study was selected for each receptor species based on study design, study methodology, study duration, study endpoint, and test species. No Observed Adverse Effect Levels (NOAELs) based on growth and reproduction were utilized, where available, as the screening values. When chronic NOAEL values were

unavailable, estimates were derived or extrapolated from chronic Lowest Observed Adverse Effect Levels (LOAELs) or acute values as follows:

- When values for chronic toxicity were not available, the median lethal dose (LD<sub>50</sub>) was used. An uncertainty factor of 100 was used to convert the acute LD<sub>50</sub> to a chronic NOAEL (i.e., the LD<sub>50</sub> was multiplied by 0.01 to obtain the chronic NOAEL).
- An uncertainty factor of 10 was used to convert a reported LOAEL to a NOAEL (USEPA, 1997b).

Ingestion screening values for mammals and birds are summarized in Tables 8-5 and 8-6, respectively. In the SERA, NOAELs were not adjusted to reflect differences in body weights between the test species and receptor species selected for evaluation.

### **8.2.3 Screening-Level Exposure Estimate**

Chemical concentrations in groundwater, surface water, surface sediment, subsurface sediment, surface soil, and subsurface soil were used to conservatively estimate potential chemical exposures for the ecological receptors selected to represent the assessment endpoints at Site 1. For conservatism, the maximum detected concentration of each compound was used in all calculations. Chemicals that were analyzed for, but not detected, were not included in exposure calculations. This decision is based on a previous agreement among the Naval Weapons Station Yorktown (WPNSTA) Partnering Team for ERAs conducted at WPNSTA and CAX.

It is noted that the use of maximum detected concentrations in groundwater is a highly conservative measure that does not account for dilution or natural attenuation of contaminants, which would likely occur during migration to a surface water body. This would tend to result in an overestimation of potential risks from groundwater.

Exposures for upper trophic level receptor species via the food web were determined by estimating the chemical-specific concentrations in each dietary component using uptake and food web models. Maximum surface soil, surface sediment, and/or surface water concentrations were used in all calculations to provide a conservative assessment. For aquatic receptors, sediment samples were not categorized by salinity as they were for lower trophic level exposure estimations. All surface sediments collected from the creek adjacent to the landfill were evaluated

together for upper trophic level exposure estimations. The approach is based on the assumption that upper trophic level receptors would not discriminate between the freshwater and saltwater portions of the tributary for foraging. Likewise, freshwater and saltwater surface water data were combined for direct contact exposure estimation for upper trophic level receptors. Only the data from the freshwater portions of the creek were used for drinking water exposure estimations. Total (unfiltered) metals data were used to evaluate drinking water exposures. Incidental ingestion of soil or sediment was also included when calculating the total level of exposure.

Subsurface soil and sediment data were not included in the food web evaluation. Since receptors (and their prey) are not exposed directly to groundwater, food web exposures were not calculated based on groundwater concentrations.

Not all chemicals were evaluated for food web exposures; only chemicals identified as "important bioaccumulative compounds" by the USEPA (USEPA, 2000) were evaluated. These chemicals are identified on Table 8-7. It should be noted that not all compounds identified as important bioaccumulative compounds by the USEPA were analyzed for in media at Site 1. It should further be noted that if a compound was not identified as an important bioaccumulative chemical by the USEPA, that compound was automatically excluded from the list of ecological COPCs for food web exposures.

Dietary items for which tissue concentrations were modeled included terrestrial plants, soil invertebrates (earthworms are the standard surrogate), small mammals, aquatic plants, aquatic invertebrates, fish, and frogs. Small mammals selected as dietary components for terrestrial carnivores were the meadow vole (herbivore), white-footed mouse (omnivore), and short-tailed shrew (insectivore). The methodologies used for estimation of tissue concentrations are outlined in the following section. The uptake of chemicals from the abiotic media into these food items was based (where available) on conservative (e.g., maximum or 90<sup>th</sup> percentile) BCFs or Bioaccumulation Factors (BAFs) from the literature. Default factors of 1.0 were used only when data were unavailable for chemicals in the literature.

**Terrestrial Plants.** Tissue concentrations in the aboveground vegetative portion of terrestrial plants were estimated by multiplying the maximum measured surface soil concentration of each chemical by chemical-specific soil-to-plant BCFs obtained from the literature. The BCF values used were based on root uptake from soil and on the ratio between dry-weight soil and dry-weight plant tissue. Literature values based on the ratio between dry-weight soil and wet-weight plant

tissue were converted to a dry-weight basis by dividing the wet-weight BCF by the estimated solids content for terrestrial plants (15 percent [0.15]; Sample et al. 1997).

For bioaccumulative organic chemicals without literature based BCFs, soil-to-plant BCFs were estimated using the algorithm provided in Travis and Arms (1988):

$$\log B_v = 1.588 - (0.578) (\log K_{ow})$$

where:

- $B_v$  = Soil-to-plant BCF (unitless; dry weight basis)
- $K_{ow}$  = Octanol-water partitioning coefficient (unitless)

The log  $K_{ow}$  values used in the calculations were obtained mostly from USEPA (1995b and 1995c) and are listed in Table 8-8. The soil-to-plant BCFs used in the SERA are summarized in Table 8-9.

**Earthworms.** Tissue concentrations in soil invertebrates (earthworms) were estimated by multiplying the maximum measured surface soil concentration for each chemical by chemical-specific BCFs or BAFs obtained from the literature. BCFs are calculated by dividing the concentration of a chemical in the tissues of an organism by the concentration of that same chemical in the surrounding environmental medium (in this case, soil) without accounting for uptake via the diet. BAFs consider both direct exposure to soil and exposure via the diet. Since earthworms consume soil, BAFs are more appropriate values and are used in the food web models when available. BAFs based on depurated analyses (where soil was purged from the gut of the earthworm prior to analysis) are given preference over undepurated analyses when selecting BAF values since direct ingestion of soil is accounted for separately in the food web model. BAFs for earthworms were those reported in Sample et al. 1998a.

The BCF/BAF values used were based on the ratio between dry-weight soil and dry-weight earthworm tissue. Literature values based on the ratio between dry-weight soil and wet-weight earthworm tissue were converted to a dry-weight basis by dividing the wet-weight BCF/BAF by the estimated solids content for earthworms (16 percent [0.16]; USEPA 1993b). For inorganic chemicals without available measured BAFs or BCFs, an earthworm BAF of 1.0 was assumed. The soil-to-earthworm BCFs/BAFs used in the SERA are summarized in Table 8-9.

**Small Mammals.** Whole-body tissue concentrations in small mammals (shrews, voles, and mice) were estimated using one of two methodologies. For chemicals with literature-based soil-to-small mammal BAFs, the small mammal tissue concentration was obtained by multiplying the maximum measured surface soil concentration for each chemical by a chemical-specific soil-to-small mammal BAF. The BAF values used were based on the ratio between dry-weight soil and whole-body dry-weight tissue. Literature values based on the ratio between dry-weight soil and wet-weight tissue were converted to a dry-weight basis by dividing the wet-weight BAF by the estimated solids content for small mammals (32 percent [0.32]; USEPA 1993b). BAFs for shrews were those reported in Sample et al. (1998b) for insectivores (or for general small mammals if insectivore values were unavailable), for voles were those reported for herbivores, and for mice were those reported for omnivores. The soil-to-small mammal BAFs used in the SERA are shown in Table 8-10.

For chemicals without soil-to-small mammal BAF values, an alternate approach was used to estimate whole-body tissue concentrations. Because most chemical exposure for these small mammal species is via the diet, it was assumed that the concentration of each chemical in the small mammal's tissues was equal to the chemical concentration in its diet, that is, a diet to whole-body BAF (wet-weight basis) of one was assumed. The use of a *diet* to whole-body BAF of one is likely to result in a conservative estimate of chemical concentrations for chemicals that are not known to biomagnify in terrestrial food chains (e.g., aluminum). For chemicals that are known to biomagnify (e.g., PCBs), a diet to whole-body BAF value of one will likely result in a realistic estimate of tissue concentrations based on reported literature values. For example, a maximum BAF (wet weight) value of 1.0 was reported by Simmons and McKee (1992) for PCBs based on laboratory studies with white-footed mice. Menzie et al. (1992) reported BAF values (wet-weight) for dichlorodiphenyltrichloroethane (DDT) of 0.3 for voles and 0.2 for short-tailed shrews. Resulting tissue concentrations (wet-weight) were then converted to dry weight using an estimated solids content of 32 percent (see above).

**Aquatic Plants.** Tissue concentrations in the aboveground vegetative portion of aquatic plants were estimated using the same methodologies as described above for terrestrial plants except that maximum sediment (not soil) concentrations were used in the calculation. Conservative sediment-to-aquatic plant BAFs using in the SERA are summarized in Table 8-11.

**Aquatic Invertebrates.** Tissue concentrations in aquatic invertebrates were estimated by multiplying the maximum measured sediment concentration for each chemical by chemical-specific sediment-to-invertebrate BAFs obtained from the literature. The BAF values used were based on the ratio between dry-weight sediment and dry-weight invertebrate tissue. BAFs based on depurated analyses (where sediment was purged from the gut of the organism prior to analysis) were given preference over undepurated analyses when selecting BAF values since direct ingestion of sediment is accounted for separately in the food web model.

Literature values based on the ratio between dry-weight sediment and wet-weight invertebrate tissue were converted to a dry-weight basis by dividing the wet-weight BAF by the estimated solids content for aquatic invertebrates (21 percent [0.21]; USEPA 1993b). For chemicals without literature based sediment-to-invertebrate BAFs, a BAF of 1.0 was assumed. The sediment-to-invertebrate BAFs used in the SERA are summarized in Table 8-12.

**Fish.** Tissue concentrations in whole-body fish were estimated by multiplying the maximum measured sediment concentration for each chemical by chemical-specific sediment-to-fish BAFs obtained from the literature. The BAF values used were based on the ratio between dry-weight sediment and dry-weight fish tissue. Literature values based on the ratio between dry-weight sediment and wet-weight fish tissue were converted to a dry-weight basis by dividing the wet-weight BAF by the estimated solids content for aquatic invertebrates (25 percent [0.25]; USEPA 1993b). For chemicals without literature based sediment-to-fish BAFs, a BAF of 1.0 was assumed. The sediment-to-fish BAFs used in the SERA are summarized in Table 8-12.

**Frogs.** Tissue concentrations in whole-body frogs were estimated by multiplying the maximum measured sediment concentration for each chemical by the chemical-specific sediment-to-frog BAFs obtained from the literature. The BAF values used were based on the ratio between dry-weight sediment and dry-weight frog tissue. Literature values based on the ratio between dry-weight sediment and wet-weight frog tissue were converted to a dry-weight basis by dividing the wet-weight BAF by the estimated solids content for aquatic invertebrates (25 percent [0.25]; USEPA 1993b). For chemicals without literature based sediment-to-frog BAFs, a BAF of 1.0 was assumed. The sediment-to-frog BAFs used in the SERA are summarized in Table 8-11.

Dietary intakes for each upper trophic level receptor species were calculated using the following formula modified from USEPA (1993b):

$$DL_x = \frac{\left[ \sum_i (FIR)(FC_{xi})(PDF_i) \right] + [(FIR)(SC_x)(PDS)] + [(WIR)(WC_x)]}{BW} * [AUF]$$

where:

- DI<sub>x</sub> = Dietary intake for chemical x (mg chemical/kg body weight/day)
- FIR = Food ingestion rate (kg/day, dry-weight)
- FC<sub>xi</sub> = Concentration of chemical x in food item I (mg/kg, dry weight)
- PDF<sub>i</sub> = Proportion of diet composed of food item I (dry weight basis)
- SC<sub>x</sub> = Concentration of chemical x in soil/sediment (mg/kg, dry weight)
- PDS = Proportion of diet composed of soil/sediment (dry weight basis)
- WIR = Water ingestion rate (L/day)
- WC<sub>x</sub> = Concentration of chemical x in water (mg/L)
- BW = Body weight (kg, wet weight)
- AUF = Area Use Factor (unitless) = 1.0 in SERA

Life history information and exposure parameters for each receptor are summarized in Table 8-13. Dietary compositions are provided in Table 8-14. As evidenced by Table 8-13, conservative receptor-specific exposure parameters were used in the evaluation of food web exposures (maximum water ingestion rate, maximum food ingestion rates, and minimum body weights). For the SERA, an AUF of 1.0 was assumed (i.e., each receptor spends 100 percent of its time within a given area of ecological concern). As such, receptor-specific home ranges were not considered in the estimation of dietary intakes.

#### 8.4 Screening-Level Risk Calculation

the screening-level risk calculation is the final step of the SERA. In this step, maximum chemical concentrations in abiotic media or maximum exposure doses (for upper trophic level receptor species) were compared with the corresponding screening values to derive screening risk estimates. The outcome of this step is a list of preliminary ecological COPCs for each media-pathway-receptor combination evaluated or a conclusion of negligible risk.

#### **8.4.1 Selection of Preliminary Ecological Chemicals of Potential Concern (COPCs) { TC “1.3.4.1 Selection of Ecological Chemicals of Potential Concern (ECOPCs)” of C \ “4” }**

Preliminary ecological COPCs were selected using the ecological hazard quotient (HQ) method. HQs are calculated by dividing the maximum chemical concentration in the medium being evaluated by the corresponding medium-specific screening value or, in the case of upper trophic level receptors, by dividing the maximum exposure dose by the corresponding ingestion-based screening value.

##### **8.4.1.1 Abiotic Media**

For the medium-specific screening (groundwater, surface soil, surface water and sediment), chemicals with HQs greater than or equal to 1.0 were identified as preliminary ecological COPCs for the appropriate receptor groups. For a given medium, chemicals that were not detected in any samples were not evaluated in the SERA, nor were they retained as preliminary ecological COPCs for that medium. As was previously discussed, this decision was based on an agreement among the WPNSTA Partnering Team. The uncertainty that this contributes to the risk assessment is discussed in the Section 8.6 of this report. Detected chemicals without medium-specific screening values for a particular chemical are noted in the site-specific screening tables and were retained as preliminary ecological COPCs in that medium. The uncertainty that this contributes to the risk assessment is also discussed in Section 8.6.

##### **8.4.1.2 Food Web Exposures**

To select preliminary ecological COPCs by evaluating food web exposures, the maximum detected chemical concentration in each media (surface soil, surface sediment, and/or surface water) was used to estimate dietary doses for each receptor. For conservatism, exposures were based on maximum ingestion rates and minimum body weights for each receptor. It was additionally assumed that chemicals were 100 percent bioavailable to the receptor and that each receptor spent 100 percent of its time on the site (i.e. an Area Use Factor of 1.0 was assumed). All detected chemicals identified as important bioaccumulative chemicals by the USEPA (2000) were evaluated in the food web model. HQs are calculated with NOAELs, LOAELs, and Maximum Acceptable Toxicant Concentrations (MATCs) (the geometric mean of the NOAEL and LOAEL). Calculations with NOAELs provided the most conservative risk estimate, while calculations with LOAELs provided the least conservative risk estimate. Calculations with MATCs provided

realistic risk estimates since the MATC represents an estimation of the threshold concentration (i.e., the concentration above which a toxic effect on the test endpoint is produced). For the SERA, chemicals with NOAEL-based HQs greater than or equal to 1.0 were considered preliminary ecological COPCs. Identical to the media-specific screening, chemicals without ingestion-based screening values were retained as preliminary ecological COPCs for upper trophic level receptors and are identified within the upper trophic level screening tables.

HQs equal to or exceeding one indicate the potential for risk since the chemical concentration or dose (exposure) exceeds the screening value (effect). However, screening values and exposure doses were derived using intentionally conservative assumptions (e.g., maximum media concentrations, maximum ingestion rates, and minimum body weights) such that HQs greater than or equal to one do not necessarily indicate that risks are present or impacts are occurring. Rather, they identify chemical-pathway-receptor combinations requiring further evaluation. Following the same reasoning, HQs less than one indicated that risks are very unlikely, enabling a conclusion of no unacceptable risk to be reached with high confidence.

It is noted that this risk assessment considers independent effects of detected compounds. However, the potential does exist for multiple contaminants in environmental media to interact. Much uncertainty is involved with the interpretation of chemical interactions due to the complexity of potential effects (e.g., synergistic, antagonistic, or additive), and due to varying toxicities of compounds in different species. For these reasons, cumulative effects are not addressed in the SERA.

#### **8.4.2 Results of SERA**

Results of the groundwater, soil, surface water, sediment, and food web HQ calculations are presented in Tables 8-15 to 8-25. Detected chemicals with HQs  $\geq 1.0$ , as well as detected chemicals lacking media-specific and/or ingestion-based screening values, were identified as preliminary ecological COPCs in this evaluation.

##### **8.4.2.1 Groundwater**

Groundwater data from four monitoring wells sampled in 2000 were screened against freshwater surface water screening values and are presented in Table 8-15. Three VOCs, one explosive, and fourteen inorganic compounds were detected in Site 1 groundwater. Only di-n-octylphthalate (HQ

= 1.33), aluminum (HQ = 3.75), iron (HQ = 93.13), and manganese (HQ = 4.21) were detected at concentrations exceeding screening values. A screening value was not available for the explosive 4-nitrotoluene. As surface water data was available for the SERA, the groundwater data were not used to determine potential risks to ecological receptors.

#### 8.4.2.2 Soils

The comparison of surface soil data (0-6 inches bgs) from Site 1 to soil screening values is presented in Table 8-16. Three VOCs, twenty-two SVOCs, eight pesticides, two PCB congeners, and twenty-three inorganics were detected in surface soils. Acenaphthylene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, pyrene, 4,4'-Dichlorodipenyldichloroethylene (4,4'-DDE), 4,4'-DDT, dieldrin, Aroclor-1248, Aroclor-1260, aluminum, antimony, cadmium, chromium, copper, cyanide, iron, lead, mercury, silver, vanadium and zinc were all detected at concentrations exceeding screening values and were retained as preliminary ecological COPCs. HQs ranged from 1.5 (acenaphthylene) to 19.0 (benzo(b)fluoranthene) for SVOCs, from 1.2 (dieldrin) to 3.3 (4,4'-DDT) for the pesticides and from 1.1 (silver) to 159.5 (iron) for the inorganic chemicals. The PCBs Aroclor-1248 and Aroclor-1260 had HQs of 4.4 and 18.0, respectively. Methyl acetate, 2-methylnaphthalene, benzaldehyde, bis(2-ethylhexyl)phthalate, caprolactam, carbazole, and dibenzofuran were also retained as preliminary ecological COPCs as no screening values were available. It is noted that phthalate esters including bis(2-ethylhexyl)phthalate are common laboratory contaminants (USEPA 1989).

The comparison of subsurface soils data (6-24 inches bgs) collected from the drainage swale on the southern edge of the landfill to soil screening values is presented in Table 8-17. One VOC, fifteen SVOCs, seven pesticides, the PCB congener Aroclor-1260, and nineteen metals were detected. Anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, fluoranthene, indeno(1,2,3-cd)pyrene, phenanthrene, pyrene, aluminum, chromium, iron, lead, nickel, vanadium and zinc were all detected at concentrations exceeding screening values and were retained as preliminary ecological COPCs. HQs ranged from 1.0 (anthracene and benzo(k)fluoranthene) to 5.7 (fluoranthene) for SVOCs, and from 3.16 (lead) to 126 (chromium) for the inorganic chemicals. Methyl acetate, carbazole, di-n-octylphthalate, and endosulfan I were also retained as preliminary ecological COPCs as no

screening values were available. It is again noted that phthalate esters including di-n-octylphthalate are common laboratory contaminants (USEPA 1989).

#### 8.4.2.3 Freshwater Surface Water and Sediment

The comparison of surface water data from the freshwater sections of the tributary bordering the landfill to freshwater surface water screening values is presented in Table 8-18. The SVOC bis(2-ethylhexyl)phthalate and seventeen inorganics were detected. Only aluminum (HQ = 11.43), iron (HQ = 17.84), and manganese (HQ = 3.03) were detected at concentrations exceeding screening values.

Surface sediment data from the samples located in the freshwater sections of the tributary bordering the landfill were compared to freshwater sediment screening values and are presented in Table 8-19. Two VOCs, fifteen SVOCs, eleven pesticides, the PCB Aroclor-1260, and twenty-one inorganics were detected. Anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, phenanthrene, pyrene, 4,4'-Dichlorodiphenyldichloroethane (4,4'-DDD), 4,4'-DDE, 4,4'-DDT, alpha-chlordane, dieldrin, Aroclor-1016, gamma-chlordane, heptachlor, Aroclor-1260, arsenic, cadmium, copper, lead, mercury, nickel, silver, and zinc were all detected at concentrations exceeding screening values and were retained as preliminary ecological COPCs. HQs ranged from 1.42 (benzo(g,h,i)perylene) to 9.20 (benzo(a)anthracene) for the SVOCs, from 2.45 (dieldrin) to 68.18 (4,4'-DDE) for the pesticides, and from 1.55 (nickel) to 7.77 (lead) for the inorganics. The PCB Aroclor-1260 had an HQ of 12.78. The VOCs 2-butanone and chloromethane, the SVOCs benzaldehyde, caprolactam, and carbazole, the pesticides endosulfan sulfate and endrin aldehyde, and beryllium were also retained as preliminary ecological COPCS for surface sediments, as no screening values were available.

The comparison of subsurface sediment data to freshwater sediment screening values is presented in Table 8-20. Two VOCs, seventeen SVOCs, ten pesticides, the PCB Aroclor-1260, and twenty-one inorganics were detected. Acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, fluoranthene, fluorene, phenanthrene, pyrene, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, alpha-chlordane, beta-benzene hexachloride (beta-BHC), dieldrin, gamma-chlordane, heptachlor, Aroclor-1260, arsenic, cadmium, copper, lead, mercury, nickel, selenium, silver and zinc were each detected at concentrations exceeding their freshwater sediment screening value. HQs ranged from 1.37 (benzo(a)pyrene) to 5.16 (fluorene) for the

SVOCs, from 1.36 (beta-BHC) to 195 (4,4'-DDE) for the pesticides, 79.30 for the PCB Aroclor-1260, and from 1.14 (nickel) to 14.35 (lead) for the inorganics. Due to a lack of screening values, 2-butanone, chlormethane, benzaldehyde, carbazole, delta-BHC, endrin aldehyde, and beryllium were also retained as preliminary ecological COPCs. 2-butanone is considered a common laboratory contaminant by EPA, however (USEPA 1989).

#### 8.4.2.4 Saltwater Surface Water and Sediments

The comparison of surface water data from the samples located in the saltwater sections of the tributary to saltwater surface water screening values is presented in Table 8-21. The VOC methyl acetate, the SVOC bis(2-ethylhexyl)phthalate and fifteen inorganics were detected. Manganese (HQ = 26.2) was detected at concentrations exceeding screening values. Methyl acetate, aluminum, cobalt, and iron were also retained as preliminary ecological COPCs, as screening values were not available for these compounds.

Surface sediment data from the samples located in the saltwater reaches of the tributary were compared to saltwater sediment screening values and are presented in Table 8-22. Five VOCs, seventeen SVOCs, ten pesticides, two PCBs, and twenty-three inorganic compounds were detected. Acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, bis(2-ethylhexyl)phthalate, butylbenzylphthalate, chrysene, dibenz(a,h)anthracene, fluoranthene, indeno(1,2,3-cd) pyrene, phenanthrene, pyrene, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, dieldrin, gamma-BHC, gamma-chlordane, heptachlor epoxide, Aroclor-1254, Aroclor-1260, arsenic, barium, cadmium, copper, cyanide, lead, mercury, nickel, and zinc were each detected at concentrations exceeding their freshwater sediment screening value. HQs ranged from 1.00 (benzo(k)fluoranthene) to 14.58 (chrysene) for the SVOCs, from 5.17 (dieldrin) to 36.82 (4,4'-DDE) for the pesticides, from 32.60 (Aroclor-1254) to 35.68 (Aroclor-1260) for the PCBs, and from 1.79 (barium) to 13.30 (lead) for the inorganics. Acetone, 2-butanone, carbon disulfide, chloromethane, methyl acetate, benzaldehyde, carbazole, endosulfan sulfate, endrin aldehyde, endrin ketone, and beryllium were also retained as preliminary ecological COPCs, as screening values were not available.

Subsurface sediment data comparisons to saltwater sediment screening values are presented in Table 8-23. Four VOCs, fourteen SVOCs, five pesticides, two PCB congeners, and twenty-two inorganic compounds were detected in subsurface sediments. Anthracene, benzo(a)anthracene, benzo(a)pyrene, chrysene, dibenzo(a,h)anthracene, fluoranthene, phenanthrene, pyrene, 4,4'-

DDE, dieldrin, alpha-chlordane, gamma-chlordane, Aroclor-1254, Aroclor-1260, arsenic, barium, cadmium, chromium, cobalt, copper, lead, manganese, mercury, nickel, selenium, and zinc were detected at concentrations exceeding screening values and were retained as preliminary ecological COPCs. HQs ranged from 1.00 (phenanthrene) to 2.67 (fluoranthene) for the SVOCs, from 5.31 (dieldrin) to 100 (4,4'-DDE) for the pesticides, from 23.35 (Aroclor-1254) to 101 (Aroclor-1260) for the PCBs, and from 1.46 (cobalt) to 34.26 (lead) for the inorganics. The VOCs 2-butanone, carbon disulfide, chloromethane, cis-1,2-dichloroethene, the PAH carbazole, and beryllium were also retained as preliminary ecological COPCs, as no screening values were available. As mentioned above, EPA considers acetone and 2-butanone to be common laboratory contaminants (USEPA 1989).

#### 8.4.2.5 Food Web Exposures

Terrestrial upper trophic level receptors are potentially at risk from inorganics, pesticides and PCBs as shown on Table 8-24. No VOCs or SVOCs were retained as preliminary ecological COPCs. The inorganics arsenic, cadmium, chromium, copper, lead, mercury, selenium, and zinc each had NOAEL-based HQs greater than 1.0 for one or more receptors. NOAEL-based HQs greater than 10 were identified for cadmium (to the short-tailed shrew and the american robin), lead (to the american robin and mourning dove), mercury (to the short-tailed shrew, white-footed mouse, and meadow vole), and from zinc (to the american robin, mourning dove, and the eastern screech-owl). Calculated NOAEL-based HQs for the PCB congener Aroclor-1260 ranged from 1.02 (meadow vole) to 43.36 (short-tailed shrew). Aroclor-1248 had HQs greater than 1.0 for the american robin (3.09) and the eastern screech-owl (1.34). The pesticide dieldrin posed potential risks to the american robin (1.0), the short-tailed shrew (HQ = 4.99), and the white-footed mouse (1.11). NOAEL-based HQ's greater than 1.0 were also calculated for the pesticide 4,4-DDE to the american robin (4.57) and the eastern screech-owl (1.98). Silver was also retained as a preliminary ecological COPC as NOAELs and LOAELs were not available for upper trophic level avian receptors.

As evidenced by Table 8-25, potential risks to aquatic upper trophic level receptors at Site 1 are driven predominantly by inorganics, the pesticide 4,4'-DDE, and the PCB congeners Aroclor-1254 and Aroclor-1260. As with the terrestrial receptors, no VOCs or SVOCs were retained as preliminary ecological COPCs. Seven inorganic chemicals (arsenic, cadmium, chromium, copper, lead, mercury, and zinc) had NOAEL-based HQs exceeding 1.0 for one or more receptors. NOAEL-based HQs exceeding 100 were calculated representing the potential for risk from

mercury to the mallard (134.68). HQs greater than 10 were calculated for arsenic to the muskrat (25.44), for copper to the marsh wren (10.42), for lead to the marsh wren (56.03) and mallard (35.12), for mercury to the great blue heron (52.20) and muskrat (21.48) and for zinc to the marsh wren (99.19) and mallard (21.66). The pesticide 4,4'-DDE posed potential risks to the marsh wren (1.34) and great blue heron (1.21). The two PCB congeners posed potential risks to all of the receptors evaluated with the exception of the muskrat. HQs ranged from 1.24 (Aroclor-1254 to the mallard) to 26.59 (Aroclor-1260 to the wren). As with terrestrial receptors, silver was also retained as a preliminary ecological COPC as NOAELs and LOAELs were not available for upper trophic level avian receptors.

#### 8.4.2.6 Screening-Level Risk Assessment Conclusions

The SERA evaluated potential risks to ecological receptors at an area of ecological concern at Cheatham Annex Site. A summary of preliminary ecological COPCs identified in the SERA is provided on Table 8-26. Results of the SERA indicated that, based on a set of conservative exposure assumptions, there are multiple chemicals that may present a risk to the receptor communities evaluated in the screening assessment (communities from terrestrial habitat, freshwater habitat, and saltwater habitat). Therefore, Site 1 is recommended for additional evaluation in Step 3a of the Baseline Risk Assessment.

### 8.5 Step 3a of the Baseline Risk Assessment - Refinement of Conservative Exposure Assumptions

The SERA for Site 1 indicated that based on a set of conservative exposure assumptions, there are multiple chemicals that may present a risk to the receptor communities evaluated in the screening assessment (communities from terrestrial habitat, freshwater aquatic habitat, and saltwater aquatic habitat). Therefore, Site 1 was carried in to Step 3a of the Baseline Risk Assessment. The approach and results of this step are discussed in the following sections.

#### 8.5.1 Approach to Refined Risk Evaluation

A refinement of the conservative exposure assumptions employed in the Tier 1 Screen is warranted if the Screening-Level Risk Calculation indicates that there are chemicals at concentrations that may present a risk to receptor populations and communities at a particular site. According to Superfund guidance (USEPA, 1997b), Step 3 initiates the problem formulation

phase of the baseline ERA. Under Navy guidance (CNO 1999), the baseline ERA is defined as Tier 2, and the first activity under Tier 2 is Step 3a. In Step 3a the conservative assumptions employed in Tier 1 are refined and risk estimates are recalculated using the same conceptual model. The reevaluation may also include consideration of background data, the frequency at which chemicals were detected, and chemical bioavailability. Assumptions, parameter values, and methods that were modified for the recalculation of media-specific and food web exposure HQs are identified below, along with justification for each modification:

- Maximum chemical concentrations were replaced by average (arithmetic mean) chemical concentrations. For individual upper trophic level avian receptor species, average chemical concentrations provide a better estimate of the likely level of chemical exposure because each of the receptors would be expected to forage in several different areas of the site, and, in many cases, off-site. Average concentrations are also appropriate for evaluating impacts to *populations* of lower trophic level receptors (i.e., terrestrial invertebrates, terrestrial plants, aquatic plants, benthic invertebrates and fish). Because some of these receptors are relatively immobile, *individuals* are likely to be impacted by locations of maximum concentrations. However, evaluation of the average exposure case is more indicative of the level of impact that might be expected at the *population* level.
- Chemicals that were identified as preliminary ecological COPCs because screening values were not available were evaluated in Step 3a. These chemicals were evaluated by comparisons to screening values in the same chemical class, if available.
- Literature-based BCFs and BAFs based on, or modeled from, central tendency estimates (e.g., mean, median, midpoint) were used in place of maximum or high-end (e.g., 90th percentile) estimates for many chemicals. An assumed BCF/BAF of 1.0 was still used for those chemicals lacking a literature based BAF/BCF. A summary of the surface soil-to-terrestrial plant BCFs and surface soil-to-terrestrial invertebrate BAFs; surface soil-to-small mammal omnivore BAFs; sediment-to-benthic invertebrate and sediment-to-fish BAFs; and sediment-to-aquatic plants BCFs and sediment-to-frogs BAFs used in Step 3a are summarized in Tables 8-27 to 8-30, respectively.
- Central tendency estimates (e.g., mean, median, midpoint) for body weight and food ingestion rate (Table 8-31) were used to develop exposure estimates for upper trophic level receptors, rather than the minimum body weights and maximum food ingestion

rates used in the SERA. The use of central tendency exposure parameter estimates is more relevant because they represent the characteristics of a greater proportion of the individuals in the population. No changes were made to dietary compositions in Step 3a. As in the SERA, the evaluation of food web exposures assumed an AUF of 1.0.

- NOAELs were adjusted to reflect differences in body weights between the test species and receptor species selected for evaluation. In addition to the NOAELs used in the SERA, consideration was also given to food web exposure risk estimates based on LOAELs and MATCs.
- Consideration was given to Site 1 groundwater, soil, surface water, and sediment background concentrations from available on-site and CAX-wide background samples. Mean contaminant concentrations on site were compared to mean background concentration. Background concentrations that were greater than site concentrations were considered to indicate that contamination was not likely to be site related.
- Consideration was given to site-specific factors that can affect the bioavailability of chemicals in surface water and sediment to lower trophic level aquatic receptor groups. For surface water, consideration was given to the concentration of metals in the dissolved (unfiltered) fraction. For sediment, consideration was given to the effect TOC and acid volatile sulfide (AVS) has on the bioavailability of organic and inorganic chemicals, respectively. For example, in addition to bulk sediment quality guidelines, equilibrium partitioning (EqP)-based screening values based on site-specific values of the fraction of organic carbon ( $f_{oc}$ ) in sediment were considered. Details regarding factors that can affect bioavailability are provided at the end of this section.
- Consideration was given to the frequency of detection for individual chemicals (CNO 1999). However, given that the sample size for the media evaluated in the SERA was small, detection frequency was not an important consideration for some chemicals.

### **Factors affecting bioavailability**

#### *Inorganics*

Total sediment concentrations are usually not predictive of the bioavailability and toxicity of metals (Luoma 1983). However, similar to nonionic chemicals, metal concentrations in sediment pore water have been correlated with toxicity (Adams et al. 1985, Swartz et al. 1985, and Kemp and Swartz 1988). An important partitioning phase controlling the bioavailability and toxicity of cadmium, copper, lead, nickel, and zinc is acid volatile sulfide (AVS). AVS represents a reactive pool of solid-phase sulfide that is available to bind these metals, rendering them biologically unavailable and nontoxic to sediment-associated biota. Cadmium, copper, lead, nickel, and zinc, collectively termed simultaneously extracted metals (SEM), represent those metals that form a more stable complex with sulfide than does iron. Theoretically, mercury is also an SEM. However, the bioavailability of mercury appears to be controlled more by methylation than by AVS (USEPA 2000).

The SEM/AVS model states that if the AVS concentration is less than the concentration of SEM, toxicity will be observed. That is, if the ratio SEM/AVS is greater than 1.0, sufficient AVS is not available to bind all the SEM and sediment-associated biota may be exposed to toxic concentrations of these metals in the sediment pore water. Conversely, if the ratio SEM/AVS is less than 1.0, sufficient AVS is present to bind all SEM. The SEM theory has successfully predicted the toxicity of sediments containing cadmium and nickel (Ankley et al. 1991, Carlson et al. 1991, and Di Toro et al. 1992) and zinc and lead (Cases and Crecelius 1994). Results with copper have been mixed (Ankley et al. 1993). It is noted that the SEM/AVS theory can only be used to predict if sediments are or are not acutely toxic. The SEM/AVS theory has not been adapted to predict chronic toxicity.

## **Organics**

In Step 3a, screening values were derived using the USEPA equilibrium partitioning (EqP) approach (USEPA 1993a) for those organic chemicals lacking a literature-based toxicological benchmark. The USEPA has chosen the EqP approach for developing sediment quality criteria (or sediment screening values in the case of this ERA) for nonionic organic chemical constituents (USEPA 1993a). This approach was used to derive sediment screening values for nonionic organic chemical constituents lacking literature-based, bulk sediment screening values from the sources listed and discussed above.

There are three underlying assumptions to the derivation of threshold screening values using the EqP method. First, it is assumed that sediment toxicity correlates with the concentration of the

chemical in the sediment pore water and not the bulk sediment concentration (i.e., the pore water concentration represents the bioavailable fraction). Secondly, partitioning between sediment pore water and bulk sediment is assumed to be dependent on the organic content of the sediment with little dependence upon other chemical or physical properties. Finally, the EqP approach assumes that equilibrium has been attained between the sediment pore water concentration and the bulk sediment concentration.

The relationship between the concentration of a nonionic organic chemical in sediment pore water and bulk sediment is described by the partitioning coefficient,  $K_p$  (USEPA 1993a):

$$K_p = (CS)/(CPW)$$

Where CS is the concentration in bulk sediment and CPW is the concentration in sediment pore water. For a given organic chemical, the partition coefficient can be derived by multiplying the  $f_{oc}$  present in the sediment by the chemical's organic carbon partition coefficient ( $K_{oc}$ ) (USEPA 1993a):

$$K_p = (f_{oc})(K_{oc})$$

Combining the equations yields the following:

$$C_s = (K_{oc})(f_{oc})(CPW)$$

If the organic carbon content of the sediment is known, a site-specific sediment screening value (SSV) can be calculated for a given non-polar organic chemical by setting CPW equivalent to a conservative surface water screening value for that chemical (SWSV):

$$SSV = (K_{oc})(f_{oc})(SWSV)$$

In this equation, SSV represents the concentration of the chemical in bulk sediment that, at equilibrium, will result in a sediment pore water concentration equal to the surface water screening value. Sediment concentrations less than SSV would be protective of sediment-associated biota. The use of surface water threshold screening values (i.e., criteria and toxicological benchmarks) assumes that the sensitivities of sediment-associated biota and the species typically tested to derive surface water screening values such as fish and aquatic life

criteria and USEPA NAWQC (predominantly water column species) are similar. Furthermore, it assumes that levels of protection afforded by the surface water screening values are appropriate for sediment-associated biota. It is noted that the EqP approach can only be used if the  $f_{oc}$  in sediment is greater than 0.2 percent. At  $f_{oc}$  values less than 0.2 percent, other factors (e.g., particle size, sorption to nonorganic mineral fractions) become relatively more important (USEPA 1993a).

Although the EqP approach was developed by the USEPA for nonionic organic chemicals, this method was also used to derive sediment threshold screening values for ionic organic chemicals lacking literature-based bulk sediment toxicological benchmarks. Application of the EqP approach to ionic organic chemicals likely overestimates their pore water concentrations since adsorption mechanisms other than hydrophobicity may significantly increase the fraction of the chemical sorbed to sediment particles (Jones et al. 1997). Therefore, the EqP-based threshold screening values developed for ionic chemicals may be overly conservative. Regardless, application of the EqP approach to the development of sediment screening values for ionic chemicals is documented in the literature (USEPA 1996a and Jones et al. 1997).

The EqP-based sediment screening values summarized in Tables 8-32 to 8-35 are based on site-specific  $f_{oc}$  values. The  $K_{oc}$  values were estimated from the following equation (USEPA 1993a and 1996a):

$$\text{Log } K_{oc} = 0.00028 + (0.983)(\text{Log } K_{ow})$$

Where  $\log K_{ow}$  is the log octanol-water partition coefficient.  $\log K_{ow}$  values used to estimate  $K_{oc}$  values, as well as estimated  $K_{oc}$  values for organic chemicals analyzed in sediment are summarized in Table 8-36. Surface water screening values used in the derivation of EqP-based sediment screening values were taken from Table 8-4. It is noted that EqP-based sediment screening values could not be calculated for those organic chemicals lacking a surface water screening value.

### **8.5.2 Results of Refined Risk Evaluation**

The Tier 1 SERA for Site 1 indicated that based on a set of conservative exposure assumptions, there are multiple chemicals that may present a risk to each of the receptor communities used in the screening assessment (terrestrial plants and invertebrates, aquatic plants, benthic

invertebrates, and fish). The SERA also indicated that multiple chemicals may present risks to upper trophic level terrestrial and aquatic receptors species. Chemicals identified as preliminary ecological COPCs in the SERA for Site 1 are summarized in Table 8-26.

Only those chemicals selected as preliminary ecological COPCs based on maximum detected concentrations and maximum reporting limits are presented in the refined media-specific screening tables. Similarly, only those chemical-receptor combinations with an HQ value greater than or equal to 1.0 based on maximum detected concentrations or maximum reporting limits are shown in the refined food web exposure HQ summary tables. Detected chemicals lacking media-specific and/or ingestion-based screening values are also considered. A discussion of the results of the recalculation of risk estimates for Site 1 is presented in the sections that follow. Refined risk calculations for Site 1 groundwater, soil, surface water, sediment, and upper trophic level aquatic receptors are presented in this section.

#### 8.5.2.1 Groundwater

One SVOC, three inorganics, and one explosive were identified as preliminary ecological COPCs in the SERA based on maximum detected concentrations (see Table 8-26). The comparison of mean groundwater concentrations to surface water screening values is presented in Table 8-37. Based on the comparison, aluminum, iron, and manganese had HQs greater than one and were retained as ecological COPCs for Site 1 groundwater. The explosive 4-nitrotoluene lacks a surface water screening value and was also retained as an ecological COPC. The SVOC di-n-octylphthalate was not retained as an ecological COPC because it had a mean HQ less than one. Aluminum, iron, manganese, and 4-nitrotoluene were further evaluated by considering site history, background concentrations and/or bioavailability. Due to the low number of groundwater samples collected (four), frequency of detection was not considered when determining the need for further evaluation of ecological COPCs.

The explosive 4-nitrotoluene was detected in monitoring well 1-GW10, located nearest the debris pile at Site 1. This is the only one of the four groundwater samples in which the explosive was detected, and the detected concentration (0.12 J ug/L) was less than the detection limit of the analysis used (0.2 ug/L). 4-Nitrotoluene was not detected in the upgradient background monitoring well. Because there is no screening value for 4-nitrotoluene, potential risks to ecological receptors from this compound are unknown. However, due to its low detected concentration and considering the dilution that would occur should contaminated groundwater

discharge to a surface water body, it is unlikely that 4-nitrotoluene would pose a significant risk to ecological receptors in the aquatic habitats associated with Site 1. It is noted that surface water samples at Site 1 were not analyzed for explosive compounds. Considering these points, 4-nitrotoluene is not recommended for further ecological evaluation.

Both iron and manganese were detected in at least 3 of the four groundwater samples collected from Site 1 and, based on the history of the site, have the potential to be site related. Both the total recoverable and dissolved analytical data for Site 1 groundwater (see Section 5.5) show that the elevated mean concentration of manganese can be attributed to a single high concentration detected in sample 1-GW07-00.

A comparison of Site 1 total recoverable groundwater concentrations to total recoverable upgradient (on-site background) data is presented in Table 8-37. The mean total recoverable concentration of aluminum was lower than the mean on-site background concentration, indicating that aluminum in site groundwater is not likely to contribute ecological risks that would exceed those contributed by upgradient sources. Both iron and manganese were detected in mean concentrations greater than the mean background concentration.

The mean concentrations of aluminum, iron, and manganese in the SERA and the Step 3a screen were expressed as total recoverable concentrations. Because the dissolved (filtered) fraction of metals in groundwater is the fraction most likely to migrate to surface water bodies, an evaluation of dissolved mean concentrations to screening values expressed as total concentrations provides information regarding the potential for groundwater contaminants to impact surface water receptors.

A comparison of dissolved (filtered) groundwater data to surface water screening values expressed as total concentrations is presented in Table 8-38. As evidenced by the table, iron and manganese were detected in the dissolved fraction at similar maximum and mean concentrations as in the total fraction (Table 8-37), while aluminum was not detected in the dissolved fraction at all. Iron and manganese mean HQs exceeded one in the dissolved fraction. Because aluminum was not detected in the dissolved fraction of groundwater, it is unlikely to migrate to surface water at concentrations that would pose unacceptable ecological risks.

In summary, aluminum, iron, manganese, and 4-nitrotoluene were retained as ecological COPCs from Site 1 groundwater. Because aluminum was not detected in dissolved groundwater, it is

unlikely to migrate to surface water near the site at concentrations sufficient to pose adverse ecological impacts. 4-Nitrotoluene was detected in one sample at low concentrations. Because there is no screening value for 4-nitrotoluene, potential risks to ecological receptors from this compound are unknown. However, due to its low detected concentration and considering the dilution that would occur should contaminated groundwater discharge to a surface water body, it is unlikely that 4-nitrotoluene would pose a significant risk to ecological receptors in the aquatic habitats associated with Site 1.

#### 8.5.2.2 Surface Soils

Fourteen SVOCs, three pesticides, two PCBs, and twelve inorganics were identified as preliminary ecological COPCs in the SERA based on maximum detected concentrations (see Table 8-26). HQs ranged from 1.1 (silver) to 159.50 (iron). One VOC and six SVOCs were additionally identified as preliminary ecological COPCs because surface soils screening values were not available.

The comparison of mean concentrations to surface soil screening values (SSSV) is presented in Table 8-39. Based on this comparison, the PAHs acenaphthylene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, and pyrene; the PCB Aroclor-1260; and the inorganics aluminum, chromium, copper, cyanide, iron, lead, mercury, vanadium, and zinc had HQs greater than 1.0 and were retained as ecological COPCs for Site 1 surface soils. Organic chemicals detected in Site 1 surface soil lacking surface soil screening values (methyl acetate, 2-methylnaphthalene, benzaldehyde, bis(2-ethylhexyl)phthalate, caprolactum, carbazole, and dibenzofuran) were also retained as ecological COPCs.

The VOC methyl acetate and the SVOCs 2-methylnaphthalene, benzaldehyde, caprolactum, carbazole, and dibenzofuran were retained as preliminary ecological COPCs because surface soil screening values were not available. Methyl acetate was detected in much lower concentrations (1J-3J ug/kg), however, than available surface soil screening values for other VOCs, which range from 300 to 1001 ug/kg (see Table 8-16). Similarly, the detected concentrations of benzaldehyde, carbazole, and dibenzofuran were less than available surface soil screening values for other SVOCs (which are typically around 100 ug/kg, see Table 8-16). Therefore, it is unlikely that

these chemicals are present at concentrations sufficient to pose unacceptable ecological risks to the site.

A comparison of CAX background surface soil data from soils similar to the site to Site 1 surface soil data is also presented in Table 8-39. A comparison to base-wide background concentrations provides information regarding the likelihood that contaminants are site related. Contaminants were considered site related if they were detected at concentrations greater than the mean background concentration. The comparison indicates that bis(2-ethylhexyl)phthalate is not a site-related chemical. Though mean concentrations of several PAH compounds were less than mean background concentrations based on the arithmetic mean of the non-detections, these chemicals were not dropped from further consideration as they may be site related and as no PAHs were detected in background samples.

In summary, the following ecological COPCs were identified for Site 1 surface soils: the VOC methyl acetate, the SVOCs 2-methylnaphthalene, acenaphthylene, anthracene, benzaldehyde, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, bis(2-ethylhexyl)phthalate, caprolactum, carbazole, chrysene, dibenzo(a,h)anthracene, dibenzofuran, fluoranthene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, and pyrene, the PCB Aroclor-1260, and the inorganics aluminum, chromium, copper, cyanide, iron, lead, mercury, vanadium, and zinc. The SVOCs 2-methylnaphthalene, benzaldehyde, caprolactum, carbazole, and dibenzofuran were retained as preliminary ecological COPCs because surface soil screening values were not available. Based upon a comparison to screening values for similar chemicals, methyl acetate, benzaldehyde, carbazole, and dibenzofuran are unlikely to be present at concentrations sufficient to pose unacceptable ecological risks. Bis(2-ethylhexyl)phthalate was detected at concentrations within background concentrations. It is noted that many of the SVOC compounds were detected at concentrations below the reporting limit, which adds uncertainty to the potential for risk from these chemicals.

#### 8.5.2.3 Subsurface Soils

Eleven SVOCs and six inorganics were identified in the SERA as preliminary ecological COPCs for Site 1 subsurface soils based on maximum concentrations (see Table 8-26). HQs ranged from 1.0 (anthracene) to 92.20 (aluminum). One VOC, two SVOCs, and one pesticide were additionally identified as preliminary ecological COPCs as surface soils screening values were not available.

The comparison of mean concentrations to surface soil screening values is presented in Table 8-40. Based on this comparison, the PAHs anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, fluoranthene, indeno(1,2,3-cd)pyrene, phenanthrene, and pyrene and the inorganics aluminum, chromium, iron, lead, nickel, vanadium, and zinc had HQs greater than 1.0 and were retained as ecological COPCs for Site 1 subsurface soils. The organic chemicals detected in Site 1 subsurface soil that lacked surface soil screening values (methyl acetate, carbazole, di-n-octylphthalate, and endosulfan I) were also retained as ecological COPCs.

As with surface soils, the VOC methyl acetate was detected and retained as a preliminary ecological COPC because a screening value was not available. It was detected in only one sample, however, at a concentration (2J ug/kg) lower than typical screening values found available for other VOCs (300 to 1,001 ug/kg). Similarly, the SVOCs carbazole and di-n-octylphthalate were detected in only one sample at concentrations much lower (50J ug/kg and 44J ug/kg, respectively) than available SVOC screening values (100 ug/kg)(see Table 8-40). The pesticide endosulfan I was also detected at only one location at a concentration (0.96J ug/kg) much lower than other surface soil screening values available for pesticides (which range from 100 to 10,0000 ug/kg, see Table 8-40). Therefore, these chemicals are unlikely to be present at the site at concentrations sufficient to pose unacceptable ecological risks.

A comparison of CAX background surface soil data from soils similar to the site to Site 1 surface soil data is presented in Table 8-40. Contaminants were considered to be site related if they were detected at concentrations greater than the mean background concentration. The comparison indicates that aluminum, iron, and vanadium in subsurface soils are not site-related chemicals. As with the surface soils, mean concentrations of several PAH compounds were less than the mean background concentrations based on the arithmetic mean of the non-detections. These chemicals were not dropped from further consideration because they may be site-related and because PAHs were not detected in background samples.

In summary, the following ecological COPCs were retained for Site 1 subsurface soils: the VOC methyl acetate, the PAHs anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, carbazole, chrysene, di-n-octylphthalate, fluoranthene, indeno(1,2,3-cd)pyrene, phenanthrene, and pyrene, the pesticide endosulfan I, and the inorganics aluminum, chromium, iron, lead, nickel, vanadium, and zinc. Methyl acetate,

carbazole, di-n-octylphthalate, and endosulfan I were retained as preliminary ecological COPCs because surface soil screening values were not available. Based upon a comparison to screening values for similar chemicals, these four chemicals are unlikely to be present at concentrations sufficient to pose unacceptable ecological risks. Aluminum, iron, and vanadium were detected at concentrations within background concentrations. It is noted that, as with surface soils, many of the PAHs were detected at concentrations below the reporting limits, adding uncertainty to this analysis.

#### 8.5.2.4 Freshwater Surface Water

The inorganics, aluminum, iron, and manganese were identified as preliminary ecological COPCs in the Tier I screen for the freshwater sections of the unnamed tributary bordering the Site 1 landfill (see Table 8-26). The comparison of total recoverable mean concentrations to freshwater surface water screening values and background concentrations is presented in Table 8-41. Based on this comparison, each of these inorganics remains as an ecological COPC for the site. All three of these metals were detected at concentrations exceeding concentrations in the upgradient sample (site-background). None of the three ecological COPCs were detected in the dissolved fraction of surface water (Table 8-42); however, because these metals are not among those bioavailable in only the dissolved fraction, they are still estimated to have the potential to pose unacceptable risks to the aquatic habitat.

Based on this comparison, aluminum, iron, and manganese were identified as ecological COPCs in the freshwater section of the unnamed tributary and may pose unacceptable risks to aquatic receptors.

#### 8.5.2.5 Freshwater Surface Sediments

Based on a comparison to maximum concentrations, eleven SVOCs, seven pesticides, one PCB, and eight inorganics were retained as preliminary ecological COPCs for freshwater surface sediments in the SERA (see Table 8-26). HQs ranged from 1.42 for benzo(g,h,i)perylene to 68.18 for 4,4-DDE. In addition, two VOCs, three SVOCs, two pesticides, and one inorganic were retained because screening values were not available.

The comparison of mean concentrations to surface sediment screening values is presented in Table 8-43. Based on this comparison, the SVOCs anthracene, benzo(a)anthracene,

benzo(a)pyrene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, fluoranthene, phenanthrene and pyrene, the pesticides 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, dieldrin, and heptachlor, the PCB Aroclor-1260, and the inorganics cadmium, copper, lead, mercury, and zinc were retained as ecological COPCs. Chemicals without screening values that were also retained as ecological COPCs include 2-butanone, chloromethane, benzaldehyde, caprolactum, carbazole, endosulfan sulfate, endrin aldehyde, and beryllium.

A comparison of on-site background surface sediment concentrations is also presented in Table 8-43. As with soils, chemicals detected at concentrations less than the background concentration were not considered to be site-related. No ecological COPCs were detected at concentrations below background.

As discussed in Section 8.5.1, sediment screening values used in the refinement included both bulk sediment quality guidelines and EqP-based screening values based on an site-specific sediment  $f_{oc}$  of values (where an  $f_{oc}$  of 0.01 corresponds to 1.0 percent organic carbon). These screening values are considered realistic because they reflect site-specific sediment characteristics that can affect bioavailability and thus potential for exposure and risk. Metal concentrations in sediment pore water were additionally considered to examine the potential bioavailability of inorganic contaminants in sediments.

SEM and AVS measurements were made on sediments collected from Site 1 during the 2001 field investigation to assess the potential bioavailability of the SEM. The SEM/AVS model states that if the AVS concentration is less than the concentration of SEM, toxicity will be observed. That is, if the ratio SEM/AVS is greater than 1.0, sufficient AVS is not available to bind all the SEM and sediment-associated biota may be exposed to toxic concentrations of these metals in the sediment pore water.

Based on the comparison of mean sediment concentrations, four SEMs (cadmium, copper, lead, and zinc) were retained as ecological COPCs. The exceedences of bulk sediment screening values suggest that these inorganics may be impacting benthic invertebrates in the estuarine wetland system adjacent to Site 1. A comparison of total SEM concentrations to AVS concentrations for each individual sample collected from Site 1 during the 2001 investigation is presented in Table 8-44. Samples collected during the 1998 investigation are not shown since SEM/AVS measurements were not conducted on these samples. Although not retained as an ecological

COPC in the Step 3a screen, total SEM concentrations shown in Table 8-44 include contributions from nickel.

As evidenced by Table 8-44, the SEM/AVS ratio (0.17) based on mean total SEM and AVS concentrations (3.09 umole/g SEM and 17.98 umole/g AVS, respectively), for the six sediment samples collected indicates that SEM metals have little potential for impact at the population level. Two individual freshwater sediment samples (CX01-SD05 and CX01-SD07) had total SEM concentrations (umole/g) that exceeded the AVS concentration (umole/g), however. The SEM/AVS ratios for these samples were 17.90 and 1.01, respectively. At CX01-SD07, zinc contributed the greatest to the total SEM concentration; however, this contribution (0.96 umole/g) did not by itself exceed the AVS concentration (1.13 umole/g). At CX01-SD05, copper (0.32 umole/g), lead (0.22 umole/g), and zinc (2.63 umole/g) each exceeded the AVS concentration independently (0.18 umole/g), indicating the potential for bioavailability for each of these metals. This site may therefore represent a hotspot within the freshwater sections of the unnamed creek bordering Site 1.

The evaluation presented in the preceding paragraphs indicates that, with the exception of a single sample, the bioavailability of SEM metals in sediment collected from the freshwater areas of Site 1 is low. Though it appears that the potential risks from SEMs are limited to one sampling area, ecological COPCs cannot be eliminated from further evaluation based solely on SEM/AVS ratios because the SEM/AVS model is a predictive tool for acute toxicity only.

Table 8-45 compares mean sediment concentrations for organic chemicals retained as ecological COPCs in the Step 3a screen to EqP-based sediment screening values. EqP-based screening values were derived using the procedure presented in Section 8.5.1 using the mean organic carbon content of sediment collected from Site 1 freshwater areas during the 2001 field investigation (8.9 percent). The use of an average TOC concentration is appropriate for evaluating the level of impact that might be expected at the population level.

Every organic chemical (with a surface water screening value available) retained as an ecological COPCs in the Step 3a screen had an HQ values less than one when mean concentrations were compared to EqP-based sediment screening values. Consequently, these chemicals (2-butanone, chloromethane, anthracene, benzo(a)anthracene, benzo(a)pyrene, fluoranthene, phenanthrene, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, Aroclor-1260, dieldrin, endosulfan sulfate, endrin aldehyde, and heptachlor) are unlikely to be present in forms that are bioavailable to Site 1 ecological receptors.

Chemicals without screening values available included 2-butanone, chloromethane, benzaldehyde, benzo(k)fluoranthene, caprolactam, carbazole, chrysene, dibenzo(a,h)anthracene, and pyrene. However, benzaldehyde, caprolactam, carbazole, and dibenzo(a,h)anthracene were detected frequently below the reporting limits at concentrations within the range of available SVOC bulk sediment screening values (60-700 ug/kg, see Table 8-43) and EqP-based values (1,262 - 3,817,083 ug/kg, see Table 8-45). Therefore, it was assumed that the potential risks to aquatic plants and invertebrates from these chemicals would be minimal.

Benzo(k)fluoranthene, chrysene, and pyrene, produced HQs greater than one when compared to bulk sediment screening values and did not have screening values available for an EqP-based comparison. These chemicals were retained as ecological COPCs for Site 1 freshwater surface sediments. For each of these chemicals, however, concentrations exceeding detection limits were only measured at two sampling locations, CX01-SD08 and 1-SD02 (see Figure 8-1). These areas may represent localized hotspots that deserve further consideration.

The following compounds were identified as ecological COPCs in freshwater surface sediments of the unnamed tributary: the VOCs 2-butanone and chloromethane, the SVOCs anthracene, benzaldehyde, benzo(a)anthracene, benzo(a)pyrene, benzo(k)fluoranthene, caprolactam, carbazole, chrysene, dibenz(a,h)anthracene, fluoranthene, phenanthrene, and pyrene, the pesticides 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, dieldrin, endosulfan sulfate, endrin aldehyde, and heptachlor, the PCB Aroclor-1260, and the inorganics beryllium, cadmium, copper, lead, mercury, and zinc. An evaluation of the bioaccumulative potential of SEM metals (including cadmium, copper, lead, and zinc) indicated that, with the exception of a single sample location, the bioavailability of these metals in sediment collected from the freshwater areas of Site 1 is low. It is noted that the SEM/AVS model is a predictive tool for acute toxicity only. Based on an evaluation of the bioaccumulative potential of organic COPCs using EqP-based sediment screening values, it is unlikely that 2-butanone, chloromethane, anthracene, benzo(a)anthracene, benzo(a)pyrene, fluoranthene, phenanthrene, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, Aroclor-1260, dieldrin, endosulfan sulfate, endrin aldehyde, and heptachlor are present in forms that are bioavailable to Site 1 ecological receptors. Chemicals without screening values available included 2-butanone, chloromethane, benzaldehyde, benzo(k)fluoranthene, caprolactam, carbazole, chrysene, dibenzo(a,h)anthracene, and pyrene. Based on a comparison to screening values for similar compounds, the potential risks to aquatic plants and invertebrates from benzaldehyde, caprolactam, carbazole, and dibenzo(a,h)anthracene are likely to be minimal. The

sampling locations CX01-SD05, CX01-SD08, and 1-SD02 and may represent isolated areas of bioavailable organic and inorganic contamination and may benefit from further study, as described above.

#### 8.5.2.6 Freshwater Subsurface Sediments

Based on a comparison to maximum concentrations, eleven SVOCs, eight pesticides, the PCB Aroclor-1260, and nine inorganics were retained as preliminary ecological COPCs for freshwater subsurface sediments in the SERA (see Table 8-26). HQs ranged from 1.14 from nickel to 195 from 4,4'-DDE. Two VOCs, two SVOCs, one pesticide, and one inorganic were also retained without screening values.

The comparison of mean concentrations to bulk sediment screening values is presented in Table 8-46. Based on this comparison, the SVOCs acenaphthene, anthracene, benzo(a)anthracene, benzo(k)fluoranthene, chrysene, and dibenzo(a,h)anthracene, the pesticides 4,4'-DDE, 4,4'-DDT, dieldrin, heptachlor, alpha-chlordane, and gamma-chlordane, the PCB congener Aroclor-1260, and the inorganics arsenic, cadmium, copper, lead, mercury, and zinc were retained as ecological COPCs with HQs greater than 1.0. In addition, the VOCs 2-butanone and chloromethane, the SVOCs benzaldehyde and carbazole, the pesticides endrin aldehyde and delta-BHC, and the inorganic beryllium were retained, as screening values were not available.

None of the ecological COPCs were detected at concentrations less than the on-site background samples, indicating that they may be site-related.

As discussed in Section 8.5.1, EqP-based sediment screening values and consideration of metals concentrations in sediment pore water were utilized to examine the potential bioavailable fraction of organic and inorganic contaminants in sediments.

AVS/SEM data was not available from the subsurface samples collected from the freshwater sections of the unnamed creek bordering the site. Mean HQ values and the distribution of concentrations for the five SEM metals (cadmium, copper, lead, nickel, and zinc) in subsurface sediments paralleled those in surface sediments, however (see Tables 8-43 and 8-46). Though it is likely that the SEM metals in subsurface soils have little potential for impact at the population level, they cannot be excluded from further consideration without additional data. In summary, as refined HQs were greater than one and as they were detected at concentrations exceeding

background, cadmium, copper, lead, and zinc were each retained as ecological COPCs. The distribution of these metals at concentrations greater than screening values was limited to three of the six sampling locations: CX01-SD03, CX01-SD05, and CX01-SD08.

Table 8-47 compares mean sediment concentrations for organic chemicals retained as ecological COPCs in the Step 3a screen to EqP-based sediment screening values as described in Section 8.5.1. EqP-based screening values were derived using the mean organic carbon content of subsurface sediments collected from freshwater areas of the creek in 2001 (7.4 percent).

There were no chemicals that had an HQ greater than one when compared to EqP-based sediment screening values. Therefore, it is unlikely that 2-butanone, chloromethane, acenaphthene, anthracene, benzo(a)anthracene, fluorene, phenanthrene, 4,4'-DDE, 4,4'-DDT, Aroclor-1260, dieldrin, endrin aldehyde, heptachlor, alpha-chlordane, delta-BHC, and gamma-chlordane are present in forms that are bioavailable to Site 1 ecological receptors. Chemicals without screening values available included benzaldehyde, benzo(k)fluoranthene, carbazole, chrysene, and dibenzo(a,h)anthracene. As with the freshwater surface sediments described in Section 8.5.1, however, benzaldehyde and carbazole were each detected below the reporting limits at concentrations within or below the range of available SVOC bulk sediment screening values (16-600 ug/kg, see Table 8-46) and EqP-based values (1,612 - 439,853 ug/kg, see Table 8-47). Therefore, it was assumed that the potential risks to aquatic communities would be minimal and thus they were not retained as ecological COPCs.

Benzo(k)fluoranthene, chrysene, and dibenzo(a,h)anthracene produced HQs greater than one in the Step 3a screen and did not have screening values available for an EqP-based comparison. These chemicals were retained as ecological COPCs for Site 1 freshwater surface sediments. Concentrations exceeding detection limits were limited in areal distribution, however, to sampling location CX01-SD08 (see Figure 8-1). Combined with similar findings in the surface sediments, this pattern further highlights CX01-SD08 as a potential hotspot deserving further consideration.

The VOCs 2-butanone and chloromethane, SVOCs acenaphthene, anthracene, benzaldehyde, benzo(a)anthracene, benzo(k)fluoranthene, carbazole, chrysene, dibenzo(a,h)anthracene, fluorene, and phenanthrene, the pesticides 4,4'-DDE, 4,4'-DDT, delta-BHC, dieldrin, endrin aldehyde, heptachlor, alpha-chlordane, and gamma-chlordane, the PCB Aroclor-1260, and the inorganics arsenic, beryllium, cadmium, copper, lead, mercury, and zinc were retained as ecological COPCs

in freshwater subsurface sediments of the unnamed creek. Based on an evaluation of the bioaccumulative potential of organic COPCs using EqP-based sediment screening values, it is unlikely that 2-butanone, chloromethane, acenaphthene, anthracene, benzo(a)anthracene, fluorene, phenanthrene, 4,4'-DDE, 4,4'-DDT, Aroclor-1260, dieldrin, endrin aldehyde, heptachlor, alpha-chlordane, delta-BHC, and gamma-chlordane are present in forms that are bioavailable to Site 1 ecological receptors. Chemicals without screening values available included benzaldehyde, benzo(k)fluoranthene, carbazole, chrysene, and dibenzo(a,h)anthracene. Based on a comparison to screening values for similar compounds, benzaldehyde and carbazole are unlikely to pose significant risks to aquatic plants and invertebrates. As with surface sediment, the sampling location CX01-SD08 may represent an isolated hotspot and may benefit from further study.

#### 8.5.2.7 Saltwater Surface Water

The inorganic manganese (HQ = 26.2) was identified as a preliminary ecological COPC in the SERA for the saltwater sections of the unnamed tributary bordering the Site 1 landfill (see Table 8-26). One VOC (methyl acetate) and three inorganics (aluminum, cobalt, and iron) were also retained, as screening values were not available.

The comparison of total recoverable mean concentrations to saltwater surface water screening values is presented in Table 8-48. This comparison supports the results from the SERA. Manganese was retained with an HQ greater than one and methyl acetate, aluminum, cobalt, and iron were retained without screening values.

Methyl acetate was detected at low concentrations in surface waters and was not detected above the reporting limits or above screening values for other VOCs elsewhere at the Site. It is likely that methyl acetate is present at concentrations near zero and that risks posed to ecological receptors are minimal.

No suitable on-site or CAX background data was available to compare to site concentrations.

Manganese and cobalt were detected in the dissolved fraction of surface water (Table 8-49). Aluminum and iron were not detected the dissolved fraction; however, because these metals are not among those bioavailable in only the dissolved fraction, they are still estimated to have the potential to pose unacceptable risks to the aquatic habitat.

Manganese, methyl acetate, aluminum, cobalt, and iron were retained as ecological COPCs for surface waters in saltwater sections of the unnamed tributary bordering the Site 1 landfill. Methyl acetate, aluminum, cobalt, and iron were retained as ecological COPCs due to a lack of screening values. Based on a comparison to screening values of similar compounds, it is likely that methyl acetate is present at concentrations near zero and that risks posed to ecological receptors from this compound are minimal.

#### 8.5.2.8 Saltwater Surface Sediments

Based on the comparison to maximum concentrations in the SERA, fifteen SVOCs, seven pesticides, two PCB congeners, and nine inorganics were retained as preliminary ecological COPCs for Site 1 saltwater surface sediments (see Table 8-26). HQs ranged from 1.0 from benzo(k)fluoranthene to 36.82 from 4,4'-DDE. Five VOCs, two SVOCs, three pesticides, and one inorganic compound were also retained without screening values.

The comparison of mean concentrations to saltwater sediment screening values is presented in Table 8-50. Based on this comparison, the SVOCs acenaphthylene, anthracene, benzo(a)anthracene, benzo(a)pyrene, bis(2-ethylhexyl)phthalate, butylbenzylphthalate, chrysene, dibenz(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, phenanthrene, and pyrene, the pesticides 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, dieldrin, endrin ketone, gamma-BHC, gamma-chlordane, and heptachlor epoxide, the PCBs Aroclor-1254 and Aroclor-1260, and the inorganics arsenic, cadmium, copper, cyanide, lead, mercury, nickel, and zinc were retained as ecological COPCs. Chemicals without screening values that were also retained include the VOCs 2-butanone, acetone, carbon disulfide and methyl acetate, the SVOCs benzaldehyde and carbazole, the pesticides endosulfan sulfate, endrin aldehyde, and endrin ketone, and the inorganic beryllium.

There were no appropriate on-site or CAX background data available to compare to site concentrations.

As discussed in Section 8.5.1, EqP-based sediment screening values and consideration to metals concentrations in sediment pore water were utilized to examine the potential bioavailable fraction of organic and inorganic contaminants in sediments.

Section 8.5.1.1 describes the approach used to examine the bioavailable fraction of the five SEM metals (cadmium, copper, lead, nickel, and zinc) in sediments. Each of these metals was retained as an ecological COPC in the Step 3a screen. The exceedences of bulk sediment screening values suggest that these inorganics may be impacting benthic invertebrates in the estuarine wetland system adjacent to Site 1. A comparison of total SEM concentrations to AVS concentrations is presented in Table 8-44.

As evidenced by Table 8-44, the average SEM/AVS ratio (2.28) based on the total SEM and AVS concentrations (28.21 umole/g SEM and 12.38 umole/g AVS, respectively), for the four sediment samples collected indicates that SEM metals may have the potential for impact at the population level. This value is largely driven by two sampling points located in the wetland bordering the creek near the large debris pile on the edge of the landfill (CX01-SD10 and CX01-SD11, see Figure 8-1). At CX01-SD11, zinc contributed the greatest to the total SEM concentration. However, this contribution (2.63 umole/g) did not by itself exceed the AVS concentration (3.26 umole/g) and the total SEM/AVS ratio just exceeded one (1.14), indicating the potential for limited bioavailability of these metals in surface sediments. At CX01-SD10, copper (89.54 umole/g) and zinc (10.20 umole/g) each exceeded the AVS concentration (1.98 umole/g) independently, indicating the potential for bioavailability for each of these metals. This site may, therefore represent a hotspot within the saltwater sections of the unnamed creek which borders Site 1.

Interestingly, no metals were found at concentrations exceeding screening values at sampling location CX01-SD04, which is the farthest downgradient sampling location. This may indicate that inorganic contamination within the creek is retained and may not have the potential to pose risks to ecological receptors utilizing the York River.

Table 8-51 compares mean sediment concentration for organic chemicals retained as ecological COPCs in the Step 3a screen to EqP-based sediment screening values as described in Section 8.5.1. The mean organic carbon content of sediment collected from Site 1 saltwater surface sediments during the 2001 field investigation (14.97 percent) was used to evaluate the level of impact that might be expected at the population level.

Each organic chemical with the exception of endrin aldehyde (HQ = 3.15), that was retained as an ecological COPC in the Step 3a screen had an HQ value less than one when mean concentrations were compared to EqP-based sediment screening values. These chemicals (carbon disulfide,

chloromethane, acenaphthylene, anthracene, benzo(a)anthracene, benzo(a)pyrene, bis(2-ethylhexyl)phthalate, butylbenzylphthalate, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, phenanthrene, pyrene, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, Aroclor-1254, Aroclor-1260, dieldrin, endosulfan sulfate, endrin ketone, gamma-BHC, gamma-chlordane, and heptachlor epoxide) are unlikely to be present in forms that are bioavailable to Site 1 ecological receptors.

Chemicals without screening values available included the VOCs 2-butanone, acetone, and methyl acetate and the SVOCs benzaldehyde and carbazole. However, both 2-butanone and methyl acetate were always detected below the reporting limits and below the range of available EqP-based screening values for other VOCs (28 - 3,172 ug/kg), (see Table 8-51). The detected concentration of acetone, a common laboratory contaminant, is also within the range of available surface sediment screening values as described above. Similarly, both benzaldehyde and carbazole were detected at concentrations below the reporting limits and below bulk and EqP-based sediment screening values for other SVOC (see Tables 8-50 and 8-51). Therefore, it is likely that the potential risks to aquatic plants and invertebrates from these chemicals are minimal.

The VOCs 2-butanone, acetone, carbon disulfide, chloromethane, and methyl acetate, the SVOCs acenaphthylene, anthracene, benzaldehyde, benzo(a)anthracene, benzo(a)pyrene, bis(2-ethylhexyl)phthalate, butylbenzylphthalate, carbazole, chrysene, dibenz(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, phenanthrene, and pyrene, the pesticides 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, dieldrin, endosulfan sulfate, endrin aldehyde, endrin ketone, gamma-BHC, gamma-chlordane, and heptachlor epoxide, the PCBs Aroclor-1254 and Aroclor-1260, and the inorganics arsenic, beryllium, cadmium, copper, cyanide, lead, mercury, nickel, and zinc were retained as ecological COPCs for surface sediments in the saltwater sections of the unnamed tributary. An evaluation of the bioaccumulative potential of SEM metals (including cadmium, copper, lead, and zinc) indicated that these metals are bioavailable to aquatic receptors at the population level. It is noted that the SEM/AVS model is a predictive tool for acute toxicity only. Based on an evaluation of the bioaccumulative potential of organic COPCs using EqP-based sediment screening values, it is unlikely that carbon disulfide, chloromethane, acenaphthylene, anthracene, benzo(a)anthracene, benzo(a)pyrene, bis(2-ethylhexyl)phthalate, butylbenzylphthalate, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, phenanthrene, pyrene, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, Aroclor-1254, Aroclor-1260, dieldrin, endosulfan sulfate, endrin ketone, gamma-BHC, gamma-chlordane, and heptachlor epoxide are

present in forms that are bioavailable to Site 1 ecological receptors. Chemicals without screening values available included 2-butanone, acetone, methyl acetate, benzaldehyde, and carbazole. Based on a comparison to screening values for similar compounds, the potential risks to aquatic plants and invertebrates from these compounds are likely to be minimal. The area surrounding CX01-SD10 may represent a localized zone for bioavailable SEM metals and may benefit from further study. In addition, as metals were not found at concentrations exceeding screening values at the most downstream depositional sampling location (CX01-SD04), there may be little potential for migration of inorganic contaminants off the site and into the York River.

#### 8.5.2.9 Saltwater Subsurface Sediments

Based on a comparison to maximum concentrations, eight SVOCs, four pesticides, the PCBs Aroclor-1254 and Aroclor-1260, and twelve inorganics were retained as preliminary ecological COPCs for saltwater subsurface sediments in the SERA (see Table 8-26). HQs ranged from 1.0 from phenanthrene to 101 from Aroclor-1260. Four VOCs, one SVOC, and one inorganic were also retained as screening values were not available.

The comparison of mean concentrations to bulk sediment screening values is presented in Table 8-52. Based on this comparison, the SVOCs anthracene, benzo(a)anthracene, benzo(a)pyrene, chrysene, dibenzo(a,h)anthracene, fluoranthene, phenanthrene, and pyrene, the pesticides 4,4'-DDE, dieldrin, alpha-chlordane, and gamma-chlordane, the PCB congeners Aroclor-1254 and Aroclor-1260, and the inorganics arsenic, barium, cadmium, copper, lead, manganese, mercury, nickel, selenium, and zinc were retained as ecological COPCs with HQs greater than 1.0. In addition, the VOCs 2-butanone, carbon disulfide, chloromethane, and cis-1,2-dichloroethene, the SVOC carbazole, and the inorganic beryllium were retained, as screening values were not available.

There were no appropriate on-site or CAX background data available to compare to site concentrations.

EqP-based sediment screening values and consideration to metals concentrations in sediment pore water were utilized to examine the potential bioavailable fraction of organic and inorganic contaminants in sediments.

AVS/SEM data was not available from the subsurface samples collected from the freshwater sections of the unnamed creek bordering the site. Though slightly elevated, mean HQ values and the distribution of concentrations for the five SEM metals (cadmium, copper, lead, nickel, and zinc) in subsurface sediments paralleled those in surface sediments, however (see Tables 8-50 and 8-52). There is some potential for population-level impacts from the SEM metals in saltwater surface sediments at this site.

It should be noted that for all of the metals retained as ecological COPCs, only one (selenium), was detected at concentrations (1.3J mg/kg, within the reporting limit) exceeding the screening value (1 mg/kg) at sampling location CX01-SD04. This pattern was also identified in surface sediments from this area of the unnamed stream. This is important as it indicates that inorganic contamination of the unnamed creek could be retained within the system and may not pose potential risks to ecological receptors utilizing the York River just downstream.

The comparison of mean sediment concentrations for organic chemicals retained as ecological COPCs in the Step 3a screen to EqP-based sediment screening values is presented in Table 8-53. The mean organic carbon content of sediment collected from saltwater subsurface sediments (18.13 percent) was used to evaluate the level of impact that might be expected at the population level.

All of the organic chemicals retained as ecological COPCs in the Step 3a screen had HQ values less than one when mean concentrations were compared to EqP-based sediment screening values. These chemicals (carbon disulfide, chloromethane, cis-1,2-dichloroethene, anthracene, benzo(a)anthracene, benzo(a)pyrene, chrysene, dibenzo(a,h)anthracene, fluoranthene, phenanthrene, pyrene, 4,4'-DDE, Aroclor-1254, Aroclor-1260, dieldrin, alpha-chlordane, and gamma-chlordane) are unlikely to be present in forms that are bioavailable to Site 1 ecological receptors.

Chemicals without screening values available included the VOC 2-butanone, and the SVOC carbazole. However, both of these contaminants were always detected below the reporting limits and below the lowest available EqP-based screening values for other VOCs (34 ug/kg), and SVOCs (3,863 ug/kg) (as shown in Table 8-53). Therefore, it is likely that the potential risks to aquatic plants and invertebrates from these chemicals are minimal. In summary, no organic contaminants were retained as ecological COPCs as their bioavailability, and thus potential for risk at the population level, is considered to be minimal.

The following chemicals were identified as ecological COPCs for subsurface sediments in saltwater sections of the unnamed tributary: 2-butanone, carbon disulfide, chloromethane, cis-1,2-dichloroethene, anthracene, benzo(a)anthracene, benzo(a)pyrene, carbazole, chrysene, dibenzo(a,h)anthracene, fluoranthene, phenanthrene, pyrene, 4,4'-DDE, dieldrin, alpha-chlordane, gamma-chlordane, Aroclor-1254, Aroclor-1260, arsenic, barium, beryllium, cadmium, copper, lead, manganese, mercury, nickel, selenium, and zinc. Based on an evaluation of the bioaccumulative potential of organic COPCs using EqP-based sediment screening values, carbon disulfide, chloromethane, cis-1,2-dichloroethene, anthracene, benzo(a)anthracene, benzo(a)pyrene, chrysene, dibenzo(a,h)anthracene, fluoranthene, phenanthrene, pyrene, 4,4'-DDE, Aroclor-1254, Aroclor-1260, dieldrin, alpha-chlordane, and gamma-chlordane are unlikely to be present in forms that are bioavailable to Site 1 ecological receptors. Chemicals without screening values available included 2-butanone and carbazole. Based on a comparison to screening values for similar compounds, the potential risks to aquatic plants and invertebrates from these compounds are likely to be minimal. It is noted that of the inorganic ecological COPCs, none were detected at concentrations exceeding screening values at the most downstream depositional sampling location (CX01-SD04, see Figure 8-1). This indicates that it is likely that these contaminants are being retained on-site and are unlikely to pose risks to potential ecological receptors utilizing the nearby York River.

#### 8.5.2.10 Food Web Exposures

The refined risk calculations for upper trophic level terrestrial and aquatic receptor species are presented separately in the sections that follow.

##### *Terrestrial Exposures*

The Tier I SERA identified two pesticides, three PCB congeners, and eight inorganic compounds as preliminary ecological COPCs for terrestrial and aquatic food web exposures at Site 1 based on conservative exposure assumptions (see Table 8-26). The inorganics arsenic, cadmium, chromium, copper, lead, mercury, selenium, and zinc were identified as preliminary ecological COPCs for nearly every terrestrial receptor. The pesticides 4,4-DDE and dieldrin were identified as posing potential risks to the american robin, eastern screech-owl, short-tailed shrew, and the white-footed mouse. The PCB congeners Aroclor-1248 and 1260 were identified as preliminary

ecological COPCs for the short-tailed shrew, white-footed mouse, meadow vole, red fox, american robin, and the eastern screech-owl (see Table 8-24).

The comparison of exposure doses calculated using more realistic, less conservative exposure parameters to ingestion-based screening values for the preliminary ecological COPCs identified in the Tier I screen is presented in Table 8-54 for terrestrial receptors. As evidenced by Table 8-54, NOAEL-based HQs were greater than one for arsenic for the short-tailed shrew and for lead for the American robin. These chemicals were identified as ecological COPCs and were further evaluated by considering background and HQs based on LOAELs and MATCs. This evaluation is presented below.

The food web model for the short-tailed shrew and American robin considered incidental ingestion of soils, and the ingestion of water and food items (terrestrial plants and invertebrates) that may have potentially bioaccumulated chemicals from soils. Arsenic and lead were detected in both freshwater surface waters and surface soils.

The comparison of Site 1 surface soil data to CAX background samples is presented in Table 8-55. Concentrations of arsenic and lead were greater on site than at background locations, suggesting that the presence of these chemicals at Site 1 may be attributed to site activities. However, as evidenced by Table 8-54, HQs based on LOAELs and MATCs were less than one for each receptor-chemical combination, indicating that actual exposure impacts are likely to be minimal.

Based on an evaluation of NOAEL-based HQs, arsenic and lead were retained as ecological COPCs for the terrestrial food web evaluation. Based on the weight of evidence provided by the comparison of Site 1 data to background data and HQs based on LOAELs and MATCs, these compounds are unlikely to pose unacceptable risks to upper trophic level terrestrial receptors at Site 1.

#### *Aquatic Exposures*

The Tier I SERA identified arsenic, cadmium, chromium, copper, lead, mercury, and zinc as preliminary ecological COPCs for the majority of the aquatic receptors evaluated (see Table 8-26). The pesticide 4,4'-DDE was identified a potential risk driver to the marsh wren and to the great blue heron. The PCBs Aroclor-1254 and Aroclor-1260 were identified as preliminary

ecological COPCs for the raccoon, mink, marsh wren, great blue heron, and the mallard (see Table 8-25).

The comparison of exposure doses calculated using more realistic, less conservative exposure parameters to ingestion-based screening values for the preliminary ecological COPCs identified in the Tier I screen is presented in Table 8-56 for aquatic receptors. As evidenced by Table 8-56, NOAEL-based HQs were greater than one for arsenic to the raccoon, lead to the marsh wren, mercury to the great blue heron and the mallard, zinc to the marsh wren, and Aroclor-1260 from the great blue heron.

The foodweb model for aquatic species considered incidental ingestion of surface sediment, and the ingestion of water and food items (aquatic plants and invertebrates, fish and frogs, depending on the receptor species, see Table 8-14), which may have potentially bioaccumulated chemicals from surface waters and sediments. Aroclor-1260 was not detected in either freshwater or saltwater surface waters, but was detected in sediments from the aquatic habitat. Arsenic, lead, and zinc were detected in sediments and surface waters from both the freshwater and saltwater sections of the unnamed creek. Mercury was also not detected in surface waters collected from the freshwater sections of the unnamed creek, but was detected in the saltwater areas of the creek (see Table 8-21). Aroclor-1260, arsenic, lead, mercury, and zinc were not detected in background surface water samples.

The comparison of Site 1 surface sediment data to on-site background samples (from the freshwater habitat) is presented in Table 8-57. The maximum and mean concentrations of each chemical were significantly lower in samples collected from the background locations, suggesting a potential link to Site 1 as the source of Aroclor-1260, arsenic, lead, mercury, and zinc concentrations in surface sediments. Elevated concentrations of lead and zinc were detected at sampling locations CX01-SD03, CX01-SD08, and CX01-SD10, but they were not solely responsible for risks to the aquatic receptors identified.

As evidenced by Table 8-56, HQs for arsenic, lead, zinc, and Aroclor-1260 based on LOAELs and MATCs were less than one for each receptor-chemical combination. HQs for mercury based on MATCs and LOAELs were less than one for the mallard but were greater than one for the great blue heron (LOAEL-HQ = 2.80, MATC-HQ = 8.84).

Based on an evaluation of NOAEL-based HQ values, Aroclor-1260, arsenic, lead, mercury, and zinc were identified at ecological COPCs for upper trophic level aquatic receptors at Site 1. Based on the weight of evidence provided by the comparison of Site 1 surface water and sediment data to background data and HQs based on LOAELs and MATCs, mercury is the only compound likely to pose unacceptable ecological impacts to aquatic receptors.

## **8.6 Uncertainties and Potential Data Gaps**

Uncertainties are present in all risk assessments because of the limitations of the available data and the need to make certain assumptions and extrapolations based on incomplete information. The uncertainty in the ERA is mainly attributable to the following factors:

Analytical Data. A source of uncertainty associated with the analytical data applies to detection limits. For many chemicals, maximum reporting limits exceeded screening values in surface soil, surface water, and sediment.

Exposure point concentrations. As is typical in a SERA, a finite number of samples of environmental media are used to develop the exposure estimates. The maximum measured concentration provides a conservative estimate for immobile biota or those with a limited home range. The most realistic exposure estimates for mobile species with relatively large home ranges and for species populations (even those that are immobile or have limited home ranges) are those based on mean chemical concentration in each medium to which these receptors are exposed. This is reflected in the wildlife dietary exposure models contained in the Wildlife Exposure Factors Handbook (USEPA 1993b), which specify the use of average media concentrations. Given the mobility of the upper trophic level receptor species used in the SERA, the use of maximum chemical concentrations (rather than mean concentrations) to estimate the exposure via food webs is very conservative. The use of mean concentrations in Step 3a addresses this uncertainty and provides a more realistic estimate of exposures and effects. Home ranges, however, were not modified from an AUF of 1.0 in the refinement. This adds an additional measure of conservatism to the refinement; risks may still be overestimated for upper trophic level receptors with relatively large home ranges.

Selection of preliminary ecological COPCs. Detected chemicals without available screening values for a particular medium were identified as preliminary ecological COPCs. This likely overstates the number of actual COPCs. Many of these chemicals (2-butanone, acetone,

methylene chloride, toluene, phthalate esters) are identified by EPA as common laboratory contaminants, adding an additional source of uncertainty (USEPA 1989 - RAGS). In Step 3a, concentrations of these chemicals were compared to screening values from other chemicals in the same category (e.g., acetone was compared to the range of screening values available for other VOCs). Though this approach was conservative in nature (ranges and minimum screening values were used), it does represent an uncertainty in the analysis.

A second source of uncertainty associated with the selection of preliminary ecological COPCs is the exclusion of non-detected chemicals from the evaluation of ecological risks. If present in abiotic media at concentrations greater than media-specific screening values and/or in food items at concentrations that would result in dietary intakes greater than ingestion-based screening values, the final list of preliminary ecological COPCs may understate the actual number of COPCs.

A third source of uncertainty associated with the selection of preliminary ecological COPCs applies to the use of NOAEL-based screening values in risk calculations for upper trophic level receptors. Use of NOAEL-based screening values is extremely conservative since they give no indication as to how much higher a concentration must be before adverse effects are observed. This uncertainty was addressed in the refinement as LOAEL and MACT-based screening values were additionally considered to estimate more realistic exposures to upper trophic level receptors.

Media-specific screening values. The bulk sediment screening values that were calculated in the Tier 1 screen using 1.0 percent sediment organic carbon do not reflect the site-specific bioavailability of chemicals to ecological receptors. These factors tend to make the resulting benchmark values very conservative and likely overestimates potential risk. Sediment screening values employed in the Step 3a exposure estimate used USEPA's EqP approach and were calculated with site-specific sediment organic carbon data, representing more realistic estimates of potential risk.

Current USEPA guidance (USEPA 1995c and 1999a) indicates that the dissolved metal fraction of certain metals (arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, and zinc) more closely estimates the bioavailable fraction. For conservatism, total recoverable surface water concentrations were compared to total recoverable surface water screening values in the Tier I screen. High levels of suspended solids and sediment-adsorbed metals in surface water samples would result in overstating bioavailability and thus potential exposures and risks to

lower trophic level aquatic receptor groups. This uncertainty was addressed in Step 3a, where dissolved (bioavailable) fractions of metals were considered.

Ingestion-based screening values. Data on the toxicity of many chemicals to the receptor species were sparse or lacking, requiring the extrapolation of data from other wildlife species or from laboratory studies with non-wildlife species. This is a typical limitation for ecological risk assessments because so few wildlife species have been tested directly for most chemicals. The uncertainties associated with toxicity extrapolation were minimized through the selection of the most appropriate test species for which suitable toxicity data were available. The factors considered in selecting a test species to represent a receptor species included taxonomic relatedness, trophic level, foraging method, and similarity of diet.

A second source of uncertainty related to the derivation of ingestion-based screening values applies to metals. Most of the toxicological studies on which the ingestion-based screening values for metals were based used forms of the metal (such as salts) that have high water solubility and high bioavailability to receptors. Since the analytical samples on which site-specific exposure estimates were based measured total metal, regardless of form, and these highly bioavailable forms are expected to compose only a fraction of the total metal concentration, this is likely to result in an overestimation of potential risks for these chemicals. Dissolved metal analytical data, where available, was utilized in the refinement of the Tier I screen to address this issue.

A third source of uncertainty associated with the derivation of ingestion-based screening values concerns the use of uncertainty factors. For example, LOAELs were extrapolated to NOAELs using an uncertainty factor of ten. This approach is likely to be conservative since Dourson and Stara (1983, *cited in USEPA, 1997b*) determined that 96 percent of the chemicals included in a data review had LOAEL/NOAEL ratios of five or less. The use of an uncertainty factor of 10, although potentially conservative, also serves to counter some of the uncertainty associated with interspecies extrapolations, for which a specific uncertainty factor was not used.

Selection of Ecological Receptors. Although exposure pathways to reptiles are likely to be complete, a reptilian species was not selected as a receptor species in this ERA because the life history and toxicological database concerning the effects of chemicals on reptiles is severely limited. It was assumed that any reptiles present at Site 1 are not exposed to significantly higher concentrations of COPCs and are not more sensitive to chemicals than the other receptor species evaluated in the risk assessment. This is likely to be a reasonable assumption since the limited

available data indicate that this group is not generally more sensitive than the other vertebrate groups addressed in the ERA. This assumption was, however, a source of uncertainty in the risk assessment.

Food Web Exposure Modeling. In the SERA, chemical concentrations in terrestrial and aquatic food items (plants, earthworms, small mammal omnivores, aquatic invertebrates, and fish) were modeled from maximum measured media concentrations and were not directly measured. Mean concentrations were utilized in the refinement. The use of generic, literature-derived exposure models and bioaccumulation factors introduces some uncertainty into the resulting estimates. The values selected and the methodologies employed were intended to provide a conservative estimate of potential food web exposure concentrations.

A second source of uncertainty associated with the food web models applies to the use of default assumptions for exposure parameters such as BCFs and BAFs. Although BCFs or BAFs for many bioaccumulative chemicals were readily available from the literature and were used in the SERA and refined risk evaluation, the use of a default factor of 1.0 to estimate the concentration of some chemicals in receptor prey items is a source of uncertainty. For most chemicals, the assumption that the chemical body burden in the prey item is at the same concentration as in soil or sediment is conservative, particularly for many metals, which are known not to bioaccumulate to any significant degree.

A third source of uncertainty associated with the food web models is the use of unrealistically conservative exposure parameters in the SERA. The use of maximum ingestion food rates and minimum body weights result in a conservative estimate of exposure. Though the refinement considered mean ingestion rates and mean body weights, AUFs were assumed to equal one in both exposure estimates. This is a conservative assumption since a significant percentage of each receptor species time could be spent foraging off-site or in unimpacted areas.

AVS/SEM. SEM metals were conservatively retained for surface sediments in the refined exposure estimation though SEM/AVS ratios indicated that there was little potential for impacts from these metals on the population level. This may represent an overestimation of the risk posed to aquatic receptors from cadmium, copper, lead, nickel, and zinc.

Chemical Mixtures. Information on the ecotoxicological effects of chemical interactions is generally lacking. This could result in an underestimation of risk (if there are additive or

synergistic effects among chemicals) or an overestimation of risks (if there are antagonistic effects among chemicals).

Data Gaps. There are several areas within this analysis that would have benefited from more complete or more comprehensive data. Data on dioxins and furans was not available for exposure estimation, for example. The site was used as a disposal area for burn residues from 1951 to 1972 (Baker, 2001). Dioxins and furans are the waste products of burning chlorinated media, including polyvinyl chloride (PVCs), plastics, and bleached paper and are commonly found in bottom and fly ash in incinerators. Although subsurface soils collected in 1992 contained analytical results for dioxins (no dioxins were measured), no data is available for furans or for either chemical class from more ecologically relevant media. As many dioxins and furans are on EPA's list of bioaccumulative chemicals (USEPA, 2000), all of the potential site contaminants may have not been adequately characterized with the available data set. More recent and more comprehensive data may be required to adequately characterize the potential for risk from these compounds.

A second source of uncertainty relating to data availability includes the potential for exposure to receptors utilizing the York River. Data from the York River, downgradient of Site 1, is not available to evaluate and delineate migration off of the site. The potential for landfill debris to erode directly into the York River is minimal as the bank bordering the landfill was stabilized and revegetated in 2000. In addition, comparisons of contaminant concentrations between the most downgradient stream sample (CX01-SD04) and those further upstream tend to indicate that contaminants can be retained within the unnamed stream. Though particle size analysis indicates that this area is characterized by coarser material (see Table 8-58), it is not necessarily indicative of scouring events (that would lead to migration to the York River) given the nature of source material (Cornwallis Cave unit) at the site. This layer is exposed along the creek, especially at its headwaters near sampling location CX01-SD01 and is characterized by medium sands containing dense shell fragments with little to no organic matter. However, each sediment sample, including CX01-SD04, was preferentially collected from depositional pools within the unnamed stream and along seepages within the wetland fringe area located between the stream and the landfill. Therefore, the potential for retention of Site-related contaminants deserves further investigation as the potential for exposure to receptors utilizing the York River habitat cannot be fully explained with the existing data set. Contaminants posing potential risks to ecological receptors have been identified in the unnamed stream bordering the site and leading to the York River. The potential for risks beyond this point is unknown.

A third potential data gap in the risk assessment relates to the distribution of samples containing elevated concentrations of PCB at the site. High concentrations of the PCB Aroclor-1260 were found in a limited area of the unnamed creek (CX01-SD08, CX01-SD09, CX01-SD11, 1SD04) and in one surface soil sample (CX01-SS10). Data from the October 1998 sampling event shows that one sample, 1-HA02-00 (see Figure 5-1), shows elevated levels of this same congener in roughly the same area of the Site (4,200J ug/kg and 5,400 ug/kg in surface soils and subsurface soils, respectively). Other congeners of PCBs were found in this sampling area in elevated concentrations as well (see Section 5.5 for a data summary). The weight of evidence suggests that this area represents a hot spot for PCB contamination and that the Site as a whole does not represent a source or a risk. It is possible that PCBs from this area are only liberated from the sediments in the marsh area during high flow/scouring events. The relatively low concentration of Aroclor-1260 in the remainder of the site would support this assessment. Additional sampling may be required to delineate this area and address the potential for risk.

Comparisons to background. Two surface water/sediment samples taken from locations upgradient of the landfill were used as site-specific background samples. One sample was located in the first depositional area downgradient of the outfall pipe under the railroad tracks. This pipe may be draining the warehouse area, but the actual area of influence is unknown. Contaminants were found at concentrations exceeding screening values from these samples, indicating the potential for anthropogenic influences on the unnamed stream that are not site related. More extensive sampling of upgradient areas would be required to further delineate the inputs of non-site risks.

A second source of uncertainty relating to comparisons with background is associated with the lack of an appropriate data set from a tidal saltwater reference or upgradient area. There is the potential that some of the contaminants measured are present at concentrations that are at background levels but this cannot be evaluated without additional data collection. Because of this, the influence of site-related contaminants may be overestimated.

Finally, only one on-site upgradient monitoring well was available to compare groundwater contaminant concentrations. Background groundwater data is available from other areas of CAX and may provide a more comprehensive reference to compare with Site 1 groundwater data. Base-wide background information would not contain site-specific influences, however, and may introduce more uncertainty into the assessment.

## **8.7 Ecological Risk Assessment Conclusions**

Table 8-59 identified ecological COPCs identified at Site 1 in Step 3a. Based on the results of the assessment, ecological receptors at the site are potentially at risk from PAHs and inorganic compounds in the terrestrial and aquatic habitats and from the PCB Aroclor-1260 in the terrestrial habitat. Therefore, further evaluation of ecological risk is recommended for Site 1. Localized areas with concentrations of potentially bioavailable contaminants include CX01-SS10, CX01-SD03, CX01-SD05, CX01-SD08, CX01-SD10, and 1-SD02.

## 9.0 CONCLUSIONS AND RECOMMENDATIONS

This section presents the conclusions and recommendations developed from the data collected at Site 1.

### 9.1 Conclusions

- The landfill is approximately 1.0 acre in extent and the northern area of impacted soil occupies an area of approximately 0.3 acres. An estimated 16,400 cubic yard of various materials was buried. The main landfill body contains mostly incinerator ash, but also is reported to include empty paint cans and paint thinner cans, cartons of ether and other unspecified drugs, railroad ties, tar paper, sawdust, rags, concrete, and lumber. A smaller portion of the landfill contains mainly soil with varying amounts of debris such as concrete and glass.
- The landfill contains materials that exhibit concentrations of PAHs, pesticides, Aroclor-1260, antimony, arsenic, copper, iron, and lead that exceed screening criteria. There are no carcinogenic risks or adverse health hazards for current receptors exposed to landfill material. These receptors include the on-site adult, adolescent trespassers, and (current/future) on-site industrial/commercial workers. Carcinogenic risks were identified for future residents exposed to soil inside the landfill. The carcinogenic risks were due to PAHs and arsenic. Noncarcinogenic hazards were identified for future construction workers and future residents exposed to soil inside the landfill. The noncarcinogenic hazards were due to antimony, arsenic, copper, and iron. Adverse health effects may also occur to the future child resident from potential exposures to lead.
- There is evidence of limited impacts from the landfill native soils. Samples from native soils below and around the landfill exhibited low concentrations of PAHs. The future child resident showed a slightly elevated noncarcinogenic health hazard from exposure to soil outside the landfill. There were no unacceptable carcinogenic risks or noncarcinogenic hazards for current receptors, future residential adult, or future construction worker receptors from potential exposure to native soils outside the landfill.
- There is limited evidence that groundwater has been impacted by the landfill. Arsenic and iron were detected in downgradient groundwater samples above background and

screening criteria. There were no unacceptable carcinogenic risks or noncarcinogenic hazards to future receptors from exposure to groundwater (non-potable use).

- There is evidence of limited impacts from the landfill to the sediment, and limited evidence of impacts to surface water. Sediment samples collected adjacent to the landfill exhibited low concentrations of PAHs, pesticides, Aroclor-1260, antimony, arsenic, copper, iron, and lead. Surface water samples collected adjacent to, as well as upstream of the landfill exhibited low concentrations of antimony, arsenic, copper, iron, and lead. There were no unacceptable carcinogenic risks or noncarcinogenic hazards to current or future receptors from potential exposure to surface water or sediment.
- A Screening Level Ecological Risk Assessment (SERA) and Step 3a was conducted for Site 1. Media evaluated in the assessment include surface and subsurface soils, groundwater, and surface water, surface sediment, and subsurface sediment from both the freshwater and saltwater portions of the unnamed tributary to the York River. Ecological receptors evaluated in the assessment included terrestrial plants and invertebrates, American robin, eastern screech owl, meadow vole, mourning dove, red fox, short-tailed shrew, white footed mouse, aquatic plants, benthic invertebrates, great blue heron, mallard, marsh wren, mink, muskrat, and raccoon. Based on the refined risk calculations and considerations of background data and bioavailability, the conclusion of the ecological risk assessment is that the levels PAHs, inorganic compounds, and the PCB Aroclor-1260 in the terrestrial habitat and levels of the PAHs and inorganic compounds in the aquatic habitats of the site may pose unacceptable risk to the ecological community. Therefore, further evaluation of ecological risks is recommended for Site 1.

## **9.2 Recommendations**

- A focused feasibility study should be conducted at Site 1.
- Remediation at Site 1 should focus on the elimination of exposure of future human and ecological receptors to landfilled materials, removal of surface debris, and to prevent erosion of landfilled materials along the banks of the York River and unnamed tributary.
- The debris pile should be surveyed and inventoried. Some materials may be sold as scrap, which could offset some of the cost for removal.

- The concentration of contaminants associated with landfilled materials is low in native soils, surface water, sediment and groundwater. Additionally, there is no carcinogenic risks or noncarcinogenic hazards for current or future human receptors from potential exposure to these contaminants. However, the ecological risk assessment for Site 1 suggests further investigation of native soils surrounding the landfill, as well as the surface water and sediment of the unnamed tributary along the western perimeter of the landfill.

## 10.0 REFERENCES

Adams, W.J., R.A. Kimerle, and R.G. Mosher. 1985. p. 429 *In Aquatic toxicology and hazard assessment: Seventh symposium*. Cardwell, R.D. , R Purdy, R.C. Bahner, Eds. American Society of Testing and Materials, Philadelphia, PA.

Agency for Toxic Substances and Disease Registry (ATSDR). 1995a. *Toxicological profile for PCBs*. U.S. Public Health Service, U.S. Department of Health and Human Services, Atlanta, GA.

Agency for Toxic Substances and Disease Registry (ATSDR). 1995b. *Toxicological profile for polycyclic aromatic hydrocarbons (PAHs)*. U.S. Public Health Service, U.S. Department of Health and Human Services, Atlanta, GA.

Agency for Toxic Substances and Disease Registry (ATSDR). 1994. *Toxicological profile for 4,4'-DDT, 4,4'-DDE, and 4,4'-DDD*. May.

Agency for Toxic Substances and Disease Registry (ATSDR). 1993. *Toxicological profile for cadmium*. U.S. Public Health Service, U.S. Department of Health and Human Services, Atlanta, GA. TP-92/06.

Agency for Toxic Substances and Disease Registry (ATSDR). 1992b. *Toxicological profile for zinc*. Draft. U.S. Public Health Service, U.S. Department of Health and Human Services, Atlanta, GA.

Agency for Toxic Substances and Disease Registry (ATSDR). 1990b. *Toxicological profile for silver*. U.S. Public Health Service, U.S. Department of Health and Human Services, Atlanta, GA. TO-90/24.

Ankley, G.T., G.L. Phipps, E.N. Leonard, D.A. Benoit, V.R. Mattson, P.A. Kosian, A.M. Cotter, J.R. Dierkes, D.J. Hansen, and J.D. Mahony. 1991. Acid-volatile sulfide as a factor mediating cadmium and nickel bioavailability in contaminated sediments. *Environ. Toxicol. and Chem.* 10:1299-1307.

Ankley, G.T., V.R. Mattson, E.N. Leonard, C.W. West, and J.L. Bennett. 1993. Predicting the acute toxicity of copper in freshwater sediments: Evaluation of the role of acid-volatile sulfide. *Environ. Toxicol. Chem.* 12:315-320.

Baes, C.F., III, R.D. Scharp, A.L. Sjoreen, and R.W. Shor. 1984. *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides Through Agriculture*. ORNL 5786. Oak Ridge National Laboratory, Oak Ridge, TN.

Baker. 1997. *Final Site Screening Process Report, Sites 1, 10, and 11. Fleet and Industrial Supply Center. Cheatham Annex, Williamsburg, Virginia*. September 1997.

Baker, 1999. Field Investigation Report, Site 1 and AOC 2, Naval Weapons Station Yorktown, Yorktown, Virginia, Cheatham Annex Site. Baker Environmental, Inc. September 1999.

Baker, 2000. Final Construction Close-Out Report, Site 1 Time-Critical Removal Action, Naval Weapons Station Yorktown, Yorktown, Virginia, Cheatham Annex Site. Baker Environmental, Inc. June 2000.

Baker. 2001. *Final Installation Restoration Program Site Management Plan, Fiscal Year 2001 at Naval Weapons Station Yorktown (WPNSTA), Cheatham Annex Site (CAX)*. March 2001.

Baker Environmental (Baker). 2002. Draft Background Investigation. Naval Weapons Station Yorktown and Cheatham Annex Site. July 2002.

Bechtel Jacobs. 1998a. *Empirical Models for the Uptake of Inorganic Chemicals from Soil by Plants*. Prepared for U.S. Department of Energy. BJC/OR-133. September 1998.

Bellrose, F.C. 1980. *Ducks, Geese, and Swans of North America*. Third Edition. Stackpole Books, Harrisburg, PA. 540 pp.

Beyer, W.N. 1990. *Evaluating Soil Contamination*. U.S. Fish and Wildlife Service Biological Report 90(2). 25 pp.

Beyer, W.N., G.H. Heinz, and A.W. Redmon-Norwood. 1996. *Environmental Contaminants in Wildlife: Interpreting Tissue Concentrations*. Lewis Publishers, Boca Raton, FL. 494 pp.

Beyer, N., E. Connor, and S. Gerould. 1994. *Estimates of Soil Ingestion by Wildlife*. Patuxent Wildlife Research Center, Laurel, MD.

Beyer, W.N. and C.D. Gish. 1980. Persistence in Earthworms and Potential Hazards to Birds of Soil Applied DDT, Dieldrin, and Heptachlor. *Journal of Applied Ecology*. 17:295-307.

Beyer, W. N. and C. Stafford. 1993. Survey and Evaluation of Contaminants in Earthworms and in Soils Derived from Dredged Material at Confined Disposal Facilities in the Great Lakes Region. *Environmental Monitoring and Assessment* 24: 151-165.

Buchman, M.F. 1999. *NOAA Screening Quick Reference Tables*. NOAA HAZMAT Report 99-1, Seattle, WA. 12pp.

Butler, R.W. 1992. Great blue heron (*Ardea herodias*). *Birds of North America*. No. 25. The Academy of Natural Sciences, Philadelphia, PA and the American Ornithologists' Union, Washington, D.C.

Casas, A.M. and E.A. Crecelius. 1994. Relationship between acid volatile sulfide and the toxicity of zinc, lead, and copper in marine sediments. *Environ. Toxicol. Chem.* 13:529-536.

Carlson, A.R., G.L. Phipps, V.R. Mattson, P.A. Kosian, and A.M. Cotter. 1991. The role of acid-volatile sulfide in determining cadmium bioavailability and toxicity in freshwater sediments. *Environ. Toxicol. Chem.* 10:1309-1319.

Chief of Naval Operations (CNO). 1999. *Navy Policy for Conducting Ecological Risk Assessments*. Memorandum from Chief of Naval Operations to Commander, Naval Facilities Engineering Command. Ser N453E/9U595355. April 5, 1999.

Chiou, 1979. "A Physical Concept of Soil-Water Equilibria for Non-Ionic Compounds." *Science*. Vol. 206, pp. 831-832. Chiou, C.T., L.J. Peters, and V.H. Freed, 1979.

Conover, M.R. 1989. Potential compounds for establishing conditioned food aversions in raccoons. *Wildlife Society Bulletin*. 17:430-435.

Cope, W.G., J.G. Wiener, and R.G. Rada. 1990. Mercury Accumulation in Yellow Perch in Wisconsin Seepage Lakes: Relation to Lake Characteristics. *Environ. Toxicol. Chem.* 9:931-940.

Davison, K.L., and J.L. Sell. 1974. DDT thins shells of eggs of mallard ducks maintained on ad libitum or controlled-feeding regimes. *Arch. Environ. Contamination and Toxicology.* 2:222-232.

Di Toro, D.M., J.D. Mahony, D.J. Hansen, K.J. Scott, A.R. Carlson, and G.T. Ankley. 1992. Acid volatile sulfide predicts the acute toxicity of cadmium and nickel in sediments. *Environ. Sci. Technol.* 26:96-101.

DON, 1998. Department of the Navy (DON). Environmental Assessment for Recreational Cabins at Fleet and Industrial Supply Center Cheatham Annex. December 1998.

Dourson, M.L. and J.F. Stara. 1983. Regulatory history and experimental support of uncertainty (safety) factors. *Regulatory Toxicology and Pharmacology.* 3:224-238.

Dunning, J.B., Jr. (editor). 1993. *CRC Handbook of Avian Body Masses*. CRC Press, Boca Raton, FL. 371 pp.

Efroymson, R.A., M.E. Will, and G.W. Suter II. 1997a. *Toxicological Benchmarks for Potential Contaminants of Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Process: 1997 Revision*. Environmental Sciences Division, Oak Ridge National Laboratory. November 1997.

Efroymson, R.A., M.E. Will, G.W. Suter II and A.C. Wooten. 1997b. *Toxicological Benchmarks for Screening Contaminants of Concern on Terrestrial Plants: 1997 Revision*. Environmental Risk Assessment Program, Oak Ridge National Laboratory, Issued November 1997.

Eisler, R. 1991. Cyanide Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. U.S. Fish and Wildlife Service Biological Report 85(1.23), *Contaminant Hazard Reviews* Report No. 23. 55 pp.

Ford, 1984. Methods of Determining Relative Contaminant Mobilities and Migration Pathways Using Physical-Chemical Data. Ford and Gruba. 1984.

Franson, M.H. (ed). 1992. *Standard Methods for Examination of Water and Wastewater, 18th Edition*. American Public Health Association, Washington D.C.

Gilbert, 1987. *Statistical Methods for Environmental Pollution Monitoring*. Gilbert, Richard O. Van Nostrand Reinhold, New York, New York.

Hem, 1992. *Study and Interpretation of Chemical Characteristics of Natural Waters*. Hem, John D., USGS Water-Supply Paper 2254. 1992

Hill, E.F., R.G. Heath, J.W. Spann, and J.D. Williams. 1975. *Lethal dietary toxicities of environmental pollutants to birds*. U.S. Fish and Wildlife Service Special Scientific Report - Wildlife No. 191, Washington D.C.

Hirsch, M.P. 1998. Bioaccumulation of silver from laboratory-spiked sediments in the oligochaete (*Lumbriculus variegatus*). *Environ. Toxicol. Chem.* 17:605-609.

Howard, P.H. 1991. *Handbook of Environmental Fate and Exposure Data for Organic Chemicals, Volume III*. Lewis Publishers, Chelsea, MI.

Johnsgrad, P. A. 1988. *North American Owls: Biology and Natural History*. Smithsonian Institution Press, Washington, D.C.

Jones, D.S., G.W. Suter II, and R.N. Hull. 1997. *Toxicological benchmarks for screening contaminants of potential concern for effects on sediment-associated biota: 1997 revision*. Environmental Restoration Division, ORNL Environmental Restoration Program. ES/ER/TM-95/R4.

Kemp, P.F. and R.C. Swartz. 1988. Acute toxicity of interstitial and particle-bound cadmium to a marine infaunal amphipod. *Mar. Environ. Res.* 26:135-153.

Krantzberg, G. and D. Boyd. 1992. The Biological Significance of Contaminants in Sediment from Hamilton Harbour, Lake Ontario. *Environ. Toxicol. Chem.* 11:1527-1540.

Levey, D.J. and W.H. Karasov. 1989. Digestive responses of temperate birds switched to fruit or insect diets. *Auk*. 106:675-686.

Long, E.R. and L.G. Morgan. 1991. The Potential for Biological Effects of Sediment-Sorbed Contaminants Tested in the National Status and Trends Program. National Oceanic and Atmospheric Administration, Seattle, WA. NOAA Technical memorandum NOS OMA 52.

Long, E.R., D.D. MacDonald, S.L. Smith, and F.D. Calder. 1995. Incidence of Adverse Biological Effects Within Ranges of Chemical Concentrations in Marine and Estuarine Sediments. *Environ. Management*. 19(1):81-97.

Luoma, S.N. 1983. Bioavailability of trace metals - a review. *Science of the Total Environment*. 28:1.

Lyman, 1982. Handbook of Chemical Property Estimation Methods. Lyman, W.J., W.F. Rechl, and D.H. Rosenblatt. 1982. McGraw-Hill, Inc., New York.

Maki, A.W. and J.R. Duthie. 1978. Summary of proposed procedures for the evaluation of aquatic hazard. Pages 153-163 in Cairns, J., Jr., K.L. Dickson, and A.W. Maki (eds). *Estimating the Hazard of Chemical Substances to Aquatic Life*. ASTM STP 657.

Maruya, K.A., R.W. Risebrough, and A.J. Horne. 1997. The bioaccumulation of polynuclear aromatic hydrocarbons by benthic invertebrates in an intertidal marsh. *Environ. Toxicol. and Chem.* 16:1087-1097.

McLane, M.A.R. and L.C. Hall. 1972. DDE thins screech owl eggshells. *Bulletin of Environmental Contamination and Toxicology*. 8:65-68.

Menzie, C.A., D.E. Burmaster, J.S. Freshman, and C.A. Callahan. 1992. Assessment of methods for estimating ecological risk in the terrestrial component: A case study at the Baird & McGuire Superfund Site in Holbrook, Massachusetts. *Environ. Toxicol. and Chem.* 11:245-260.

Ministry of Housing, Spatial Planning and Environment (MHSPE). 1994. *Intervention Values*. Directorate-General for Environmental Protection, Department of Soil Protection, The Hague, Netherlands. 9 May. DBO/07494013.

NEESA, 1984. Initial Assessment Study of Fleet and Industrial Supply Center (Norfolk) Cheatham Annex and Yorktown Fuels Division. Naval Energy and Environmental Support Activity (NEESA). February 1984.

Oliver, B.G. 1987. Biouptake of chlorinated hydrocarbons from laboratory-spiked and field sediments by oligochaete worms. *Environmental Science and Technology*. 21:785-790.

Oliver, B.G. and A.J. Niimi. 1988. Trophodynamic analysis of polychlorinated biphenyl congeners and other chlorinated hydrocarbons in the Lake Ontario ecosystem. *Environ. Science and Technology*. 22:388-397.

Ontario Ministry of Environment and Energy (MOE). 1993. *Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario*. ISBN 0-7729-9248-7. 27 pp.

Palmer, R.S. 1976. *Handbook of North American Birds: Volume 1*. Yale University Press, New Haven, CT.

Pascoe, G.A., R.J. Blanchet, and G. Linder. 1996. Food Chain Analysis of Exposures and Risks to Wildlife at a Metals-Contaminated Wetland. *Archives of Environmental Contamination and Toxicology*. 30:306-318.

Patton, J.F. and M.P. Dieter. 1980. Effects of petroleum hydrocarbons on hepatic function in the duck. *Comp. Biochem. Physiol.* 65C:33-36.

Rigdon, R.H. and J. Neal. 1963. Fluorescence of chickens and eggs following the feeding of benzopyrene crystals. *Texas Reports on Biology and Medicine*. 21(4):558-566.

Robbins, C.T. 1983. *Wildlife Feeding and Nutrition*. Academic Press, New York, NY. 343 pp.

Sample, B.E., J.J. Beauchamp, R.A. Efrogmson, G.W. Suter II, and T.L. Ashwood. 1998a. *Development and Validation of Bioaccumulation Models for Earthworms*. Environmental Restoration Division, ORNL Environmental Restoration Program. ES/ER/TM-220.

Sample, B.E., J.J. Beauchamp, R.A. Efrogmson, and G.W. Suter II. 1998b. *Development and*

*Validation of Bioaccumulation Models for Small Mammals*. Environmental Restoration Division, ORNL Environmental Restoration Program. ES/ER/TM-219.

Sample, B.E., M.S. Aplin, R.A. Efroymsen, G.W. Suter, II and C.J.E. Welsh. 1997. *Methods and Tools for Estimation of the Exposure of Terrestrial Wildlife to Contaminants*. Oak Ridge National laboratory, Oak Ridge TN. ORNL/TM-13391.

Sample, B.E., D.M. Opresko and W. Suter. 1996. *Toxicological Benchmarks for Wildlife: 1996 Revision*. Health Sciences Research Division, Oak Ridge, Tennessee.

Sample, B.E. and G.W. Suter II. 1994. *Estimating Exposure of Terrestrial Wildlife to Contaminants*. Environmental Restoration Division, ORNL Environmental Restoration Program. ES/ER/TM-125.

Silva, M. and J.A. Downing. 1995. *CRC Handbook of Mammalian Body Masses*. CRC Press, Boca Raton, FL. 359 pp.

Simmons, G.J. and M.J. McKee. 1992. Alkoxyresorufin metabolism in white-footed mice at relevant environmental concentrations of Aroclor 1254. *Fundamental and Applied Toxicology*. 19:446-452.

Stickel, L.F. 1973. Pesticide residues in birds and mammals. Pages 254-312 in C.A. Edwards (ed). *Environmental Pollution by Pesticides*. Plenum Press, New York.

Suter, G.W. II. 1989. Ecological endpoints. Chapter 2 in Warren-Hicks, W., B.R. Parkhurst, and S.S. Baker, Jr. (eds). *Ecological Assessment of Hazardous Waste Sites: a Field and Laboratory Reference*. EPA/600/3-89/013.

Suter, G.W. II. 1990. Endpoints for regional ecological risk assessment. *Environmental Management*. 14:9-23.

Suter, G.W. II. 1993. *Ecological Risk Assessment*. Lewis Publishers, Chelsea, MI. 538 pp.

Suter, G.W. II and C.L. Tsao. 1996. *Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota: 1996 Revision*. Environmental Restoration Division, ORNL Environmental Restoration Program, ES/ER/TM-96/R2. 54 pp.

Swartz, R.C., G.R. Didsforth, D.W. Schults, and J.O. Lamberson. 1985. Sediment toxicity to a marine infaunal amphipod: Cadmium and its interaction with sewage sludge. *Mar. Environ. Res.* 18:133-153.

Syracuse Research Corporation (SRC). 1998. *Experimental octanol/water partition coefficient (Log P) database*. <http://esc.syrres.com/~esc1/srckowdb.htm>.

Teifke, 1973. Stratigraphic Units of the Lower Cretaceous Through Miocene Series, in Geologic Studies, Coastal Plain of Virginia. Teifke, R. H., Virginia Division of Mineral Resources, Bulletin 83, Parts 1 and 2. 1973.

Tomlinson, R.E., D.D. Dolton, R.R. George, and R.R. Mirarchi. 1994. Mourning Dove. Pages 1-26 in T.C. Tacha and C.E. Braun (eds). *Migratory Shore and Upland Game Bird Management in North America*. Int. Assoc. Fish and Wildlife Agencies, Washington, D.C.

Travis, C.C. and A.D. Arms. 1988. Bioconcentration of organics in beef, milk, and vegetation. *Environ. Science and Technology*. 22:271-274.

United States Environmental Protection Agency (USEPA), 2002a. "Risk-Based Concentration Summary Table", April 2002. Region III, Philadelphia, Pennsylvania.

USEPA, 2002b. Integrated Risk Information System. Environmental Criteria and Assessment Office, Cincinnati, Ohio.

USEPA. 2000. *Bioaccumulation Testing and Interpretation for the Purpose of Sediment Quality Assessment. Status and Needs*. Office of Water and Office of Solid Waste, Washington, D.C. EPA-823-R-00-001.

USEPA. 1999a. *National Recommended Water Quality Criteria-Correction - 1999*. EPA 822-Z-99-001 Office of Water. April 1999.

USEPA. 1999b. *Supplemental Guidance to RAGS: Region 4 Ecological Risk Assessment Bulletins*. August 1999.

USEPA, 1998a. Aerial Photographic Analysis. USN Supply Center - Cheatham Annex. Williamsburg, Virginia. United States Environmental Protection Agency, TS-PIC-980336 4S. May 1998.

USEPA, 1998b. Risk Assessment Guidance for Superfund: Volume I Human Health Evaluation Manual (Part D, Standardizing Planning, Reporting, and Review of Superfund Risk Assessments), Interim. Office of Emergency and Remedial Response. Publication 9285.7-01D. January 1998.

USEPA. 1998c. *Guidelines for Ecological Risk Assessment*. EPA/630/R-95/002B. U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, DC.

USEPA, 1997a. Health Effects Summary Tables FY 1997. Office of Solid Waste and Emergency Response, Washington, D.C. May 1995. OERR 9200.6-303(95-1).

USEPA, 1997b. Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments (Interim Final). Environmental Response Team, EPA/540/R-97-006. June 5, 1997.

USEPA, 1997c. Exposure Factors Handbook. Office of Research and Development. Washington, D.C. August 1997. EPA/600/P-95/002F.

USEPA. 1996a. *Ecotox Thresholds*. ECO Update, Volume 3, Number 2. Office of Solid Waste and Emergency Response. EPA 540/F-95/038. January 1996.

USEPA. 1996b. *Superfund chemical data matrix*. EPA/540/R-96/028.

USEPA. 1995a. *Region III - Biological Technical Assistance Group Screening Levels* (Revised). Region III, Philadelphia, Pennsylvania. August 1995.

USEPA. 1995b. *Internal Report on Summary of Measured, Calculated and Recommended log  $K_{ow}$  Values*. Environmental Research Laboratory, Athens, GA. April 10.

USEPA. 1995c. *Final Water Quality Guidance for the Great Lakes System*. 60FR15365.

USEPA, 1995d. *Assessing Dermal Exposure from Soil*. Hazardous Waste Management Division, Office of Superfund Programs. Region III, Philadelphia, Pennsylvania. December 1995.

USEPA. 1994a. *Surface Water Quality Criteria*. Federal Register. 59FR3762.

USEPA. 1994b. *Water Quality Criteria Summary*. Office of Science and Technology, Health and Ecological Criteria Division, Washington, DC.

USEPA, 1994c. *Revised Soil Lead Guidance for CERCLA Sites and Corrective Action Facilities*. OSWER Directive 9355.4-12, July 14, 1994.

USEPA, 1994d. *Guidance Manual for the Integrated Exposure Uptake Biokinetic Model for Lead in Children*. Office of Solid Waste and Emergency Response. EPA/540/R-93/081. February 1994.

USEPA. 1993a. *Technical basis for deriving sediment quality criteria for nonionic organic contaminants for the protection of benthic organisms by using equilibrium partitioning*. Office of Water, Washington, D.C. EPA-822-R-93-011.

USEPA. 1993b. *Wildlife Exposure Factors Handbook*. Office of Research and Development. Washington, D.C. EPA/600/R-93/187a.

USEPA, 1993c. *Selecting Exposure Routes and Contaminant of Concern by Risk-Based Screening*, Region III Technical Guidance Manual. Region III, Philadelphia, Pennsylvania. January 1993. EPA/903/R-93-001.

USEPA, 1993d. *Superfund's Standard Default Exposure Factors for the Central Tendency and Reasonable Maximum Exposure*. Draft. Washington, D.C. November 4, 1993.

USEPA, 1992a. *Dermal Exposure Assessment: Principles and Applications*. Interim. Office of Research and Development. Washington, D.C. January 1992. EPA/600/8-91/011B.

USEPA, 1992b. Risk Assessment Guidance for Superfund Volume I, Human Health Evaluation Manual Supplemental Guidance. Dermal Risk Assessment. Interim Guidance. Office of Emergency and Remedial Response. Washington, D.C. August 18, 1992.

USEPA, 1992c. Supplemental Guidance to RAGS: Calculating the Concentration Term. OSWER Publication Number 9285.7-081. May 1992.

USEPA. 1992d. *Framework for Ecological Risk Assessment*. U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, DC. EPA/630/R92/001.

USEPA, 1991a. Laboratory Data Validation Functional Guidelines for Evaluating Organic Analyses. United States Environmental Protection Agency. 1991.

USEPA, 1991b. Risk Assessment Guidance for Superfund Volume I. Human Health Evaluation Manual Supplemental Guidance: "Standard Default Exposure Factors" Interim Final. Office Solid Waste and Emergency Response. Washington, D.C. March 25, 1991. OSWER Directive 9285.6-03.

USEPA. 1991c. *Criteria for Choosing Indicator Species for Ecological Risk Assessments at Superfund Sites*. EPA/101/F-90/051. 51 pp.

USEPA, 1990. National Oil and Hazardous Substances Pollution Contingency Plan; Final Rule. Federal Register, 40 Code of Federal Regulations Part 300, March 8, 1990.

USEPA, 1989a. Exposure Factors Handbook. Office of Health and Environmental Assessment. Washington, D.C. July 1989. EPA/600/8-89-043.

USEPA, 1989b. Risk Assessment Guidance for Superfund Volume I. Human Health Evaluation Manual (Part A) Interim Final. Office of Solid Waste and Emergency Response. Washington, D.C. December 1989. EPA/540/1-89-002.

USGS, 1997. Geohydrology of the Shallow Aquifer System, Naval Weapons Station Yorktown, Yorktown, Virginia. US Geological Survey Water-Resources Investigations Report 97-4188. 1997.

VDMR, 1973. Geologic Studies Coastal Plain of Virginia. Bulletin 83 (Parts 1 and 2). 1973.

VWCB, 1973. Groundwater of the York-James Peninsula. Basic Data Bulletin. 1973.

Weston, 1994. Final Site Investigation for Sites 1, 10, and 11 Naval Supply Center, Cheatham Annex. Roy F. Weston, Inc. November, 1994.