

**Supplemental Remedial Investigation
Former Fire Training Area
NASA Wallops Flight Facility**

Wallops Island, Virginia



**National Aeronautics and Space Administration
Goddard Space Flight Center
Wallops Flight Facility**

May 2004

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**SUPPLEMENTAL REMEDIAL INVESTIGATION
FORMER FIRE TRAINING AREA
NASA WALLOPS FLIGHT FACILITY
WALLOPS ISLAND, VIRGINIA**

**Submitted to:
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Goddard Space Flight Center
Wallops Flight Facility
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EXECUTIVE SUMMARY

This Supplemental Remedial Investigation (RI) has been prepared for the National Aeronautics and Space Administration (NASA) under Contract Task Order (CTO) 848 issued by the Engineering Field Activity Northeast (EFANE) of the Naval Facilities Engineering Command under the Comprehensive Long-Term Environmental Action Navy (CLEAN III) contract number N62467-94-D-0888. The Supplemental RI addresses the Former Fire Training Area (FFTA) Site at the NASA Goddard Space Flight Center (GSFC) Wallops Flight Facility (WFF) located in Accomack County, Virginia. The Supplemental RI activities and this report have been prepared as part of the NASA environmental restoration program under the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA), as amended by the Superfund Amendments and Reauthorization Act (SARA) of 1986, and Virginia Administrative Code Hazardous Waste Management Regulations, 9 VAC20. The Supplemental RI was conducted in accordance with the Final Work Plan for Supplemental RI Activities approved and finalized in January 2003 (TtNUS, 2003a).

PURPOSE

The primary scope of the Supplemental RI activities was to perform further investigations and evaluations of the FFTA site to ensure that appropriate remedial alternatives are developed and evaluated. The development and evaluation of remedial alternatives require an understanding of the site characteristics, the nature and extent of contamination, and the identification of potential exposure pathways and receptors. As described in the final work plan, the scope of the Supplemental RI included the performance of additional investigations to fill data gaps and provide supplemental data to refine the risk assessments, support engineering evaluations, and provide data needed to develop remedial alternatives for the FFTA site.

The specific Supplemental RI objectives include but are not limited to the following:

- Characterization of the source area materials.
- Characterization of groundwater conditions.
- Collect data to support the evaluation of natural attenuation processes potentially occurring within the aquifer.
- Continued refinement of the Conceptual Site Model.
- Assessment of potential human health and environmental risks.

BACKGROUND

WFF is located in Accomack County on the Eastern Shore of the Commonwealth of Virginia. The facility is comprised of three separate areas; the Main Base (MB), the Mainland (ML), and Wallops Island (WI). These three areas are in close proximity to each other and total approximately 5,000 acres of landmass and 1,000 acres of marshland. The most heavily developed area is the MB (about 1,900 acres) that includes administrative and technical offices, tracking and data acquisition components, the range control center, rocket motor storage and processing facilities, research and development facilities, airfield and control tower, aircraft hangars and maintenance facilities, and Navy administration and housing areas.

The WFF is located in the Atlantic Coastal Plain Physiographic Province and is underlain by a thick sequence of approximately 7,000 feet of unconsolidated sediments that unconformably overlie crystalline bedrock. The principal water-bearing units beneath the WFF are, in stratigraphically descending order, the Pleistocene-age Columbia Group and three hydraulically isolated aquifer units within the underlying Miocene-age Yorktown Formation. The Columbia Group extends to a subsurface depth of approximately 60 feet and consists of interbedded sands, gravels, and sandy clays deposited under fluvial and marine conditions. The water table beneath the WFF typically occurs under unconfined conditions within the Columbia Group at depths of 0 to 30 feet. Groundwater flow direction within the Columbia, as is generally the case with most shallow unconfined aquifers, reportedly mimics the general surface topography. The Yorktown Formation is approximately 1,000 feet thick and consists of alternating sequences of fine- to coarse-grained glauconitic sands (that may be variably clayey, silty, or shelly) and finer-grained silts and clays. The sands comprise the dominant aquifers, and the silts and clays form aquitards that create confining conditions and that separate the Yorktown Formation into three (upper, middle, and lower) aquifers. At the WFF, the upper Yorktown aquifer generally occurs at a depth of about 50 to 100 feet below ground surface and is isolated from the overlying Columbia aquifer by a clay and silt aquitard that is approximately 20 to 40 feet thick. Aquifer tests at the WFF indicate that there is no significant vertical leakage across the confining unit separating the upper Yorktown aquifer from the overlying Columbia Group. The WFF and the surrounding communities rely on groundwater, primarily from the Yorktown formation, for drinking water supplies.

The FFTA is located on the north side of the MB adjacent to a former taxiway. The area is currently an open grass field that gently slopes to the north and northeast. Areas of higher elevation surround the FFTA. There are no surface water bodies in or immediately near the FFTA. Surface runoff within the FFTA flows to low-lying areas within the site where it either infiltrates or evaporates.

NASA began using the FFTA for fire fighting training exercises in 1965 and continued using the area until 1987. It is reported that fire fighting training, conducted twice a week during this time period, consisted of

releasing combustible material onto the ground or into an open-top tank, shallow pit, or discarded airplane body, igniting the material, and extinguishing the flames. There are no records identifying the type of materials used during these training exercises.

Environmental investigations began at the FFTA in 1986. In 1986, the Commonwealth of Virginia conducted an inspection of the FFTA and identified substances thought to be jet fuel and crank case oil in an unlined pit. The Virginia Department of Waste Management issued a removal order based on the inspection findings. NASA responded to the order by completing a soil removal in November 1986. It is reported that approximately 120 cubic yards of impacted soils were removed. The limits of the excavation were determined by visual observations. No sampling or surveying was conducted as part of the removal action.

A preliminary assessment (PA) and site investigation (SI) of the FFTA were conducted from 1989 through 1990. The studies included the performance of soil gas surveys and surface soil and groundwater sampling and analysis using field analytical techniques. The data from these investigations indicated the presence of volatile organic compounds (VOCs) in the subsurface within the former fire training area and in the downgradient groundwater.

An RI was conducted at the FFTA from 1993 through 1994 and published in 1996. Field activities included the performance of a soil gas survey, advancement of soil borings, the installation of monitoring wells, and the collection of soil (surface and subsurface) and groundwater samples. Soil gas samples collected from throughout the study area were analyzed using a field laboratory equipped with a gas chromatogram for contaminant identification. The most prevalent and highest concentration compounds identified in this survey were 1,1-dichloroethene (1,1-DCE) and 1,1,1-trichloroethane (1,1,1-TCA). The highest level of contamination was identified in the area of the former fire training pit. Eight surface soil samples were collected in the FFTA from around the area that exhibited the elevated soil gas readings. Seventeen subsurface soil samples were collected from six borings advanced in or immediately downgradient of the FFTA and 10 monitoring wells were installed at the site at the conclusion of the soil boring program. The monitoring wells were completed at three depth intervals within the shallow Columbia aquifer and were located based on projected groundwater flow direction (to the north and northeast) and proximity to the suspected source area (fire training pit area). The new and existing monitoring wells were sampled and all soil and groundwater samples were analyzed for target compound list (TCL) organic and target analyte list (TAL) metals.

Based on the RI findings it was concluded that a groundwater contaminant plume, consisting primarily of benzene, toluene and cis-1,2-dichloroethene, was emanating from the former fire training pit area and flowing to the northeast.

A supplemental groundwater sampling investigation was conducted in February 2000. The objective of the sampling event was to collect groundwater data to assess if site conditions had changed since the RI data were collected. The results of the 2000 sampling were compared to the RI findings and it was concluded that the apparent groundwater plume had diminished in the source area and broadened in the downgradient area.

Based on the results of the 1996 RI (soil gas, surface soil, subsurface soil, groundwater) and 2000 (groundwater only) sampling events, it was concluded that past fire fighting training exercises were the source of groundwater contamination identified north of the abandoned taxiway. Hydrogeologic investigations concluded that groundwater in the area occurs in an unconfined water table aquifer within the Columbia Group that is isolated, by an aquitard, from the lower Yorktown formation. Groundwater flow was projected to be in a northeasterly direction and based on the available analytical data it was suggested that contamination, primarily VOCs, was migrating with the groundwater and possibly being discharged to the area wetlands and creeks.

SUPPLEMENTAL 2003 RI FIELD ACTIVITIES AND FINDINGS

The scope of the Supplemental RI field investigations included collecting and analyzing surface and subsurface soil samples from within the suspected source area, installing additional monitoring wells, and sampling the new and existing monitoring wells to better define the groundwater flow and contaminant migration pathway.

A total of 18 surface and 37 subsurface soil samples were collected from the suspected source area and analyzed for VOCs, semi-volatile organic compounds (SVOCs), polychlorinated biphenyl (PCBs), and TAL metals. Additionally, surface soil samples were analyzed for dioxin (and furans), pH, total organic carbon (TOC), and grain size.

Seven new monitoring wells were installed at the FFTA. The purpose of the new wells was to fill data gaps to provide a more complete delineation of groundwater flow patterns (by providing additional hydraulic head data) and a more complete delineation of the nature and extent of groundwater contamination. Four shallow wells were installed across the water table and three deep wells were installed to monitor the lower portion of the Columbia aquifer. The well borings were lithologically logged to further investigate hydrogeologic conditions at the FFTA and water level measurements and groundwater samples were collected and analyzed from the new and existing wells. Ground water samples were analyzed for VOCs plus methyl tert-butyl ether (MTBE), SVOCs, PCBs, TAL total and dissolved metals (field-filtered), and monitored natural attenuation (MNA) parameters (hydrogen, soluble manganese, nitrate, chloride, sulfate, methane/ethane/ethene, and TOC. Additionally, field parameters (pH, temperature, dissolved oxygen,

oxygen reduction potential (ORP), salinity, conductivity, turbidity, alkalinity, ferrous iron, and hydrogen sulfide) were measured at each well.

The analytical data from the Supplemental RI were validated in accordance with EPA Region 3 and national functional guidelines and were evaluated along with the historical RI data to further define the nature and extent of contamination at the FFTA. In addition, the lithological logs from the new and existing wells and the hydraulic head data collected during the Supplemental RI were evaluated to further refine the conceptual site model. The results and findings of the evaluation are presented and discussed in this report. The major findings and conclusions of the Supplemental RI are as follows:

- The lithologic unit immediately underlying the site, the Columbia Group, was found to consist of predominantly fine- to medium-sand with lesser amounts of silt and clay. The Columbia Group is approximately 50 feet thick beneath the FFTA.
- Consistent with the previous findings, a silty clay layer was encountered between 47 and 52 feet below the found surface. This clay is interpreted as the upper aquitard of the Yorktown Formation based on the agreement between the resultant observed thicknesses of the Columbia Group (approximately 50 feet) with its estimated regional thickness.
- A silty clay layer, approximately 3 feet thick, exists within the Columbia Group beneath the FFTA at a subsurface elevation near sea level (25 feet below ground surface). This clay lens functions as a leaky aquitard that hydraulically divides the Columbia aquifer beneath the FFTA into upper and lower units.
- Groundwater in the upper unit of the Columbia aquifer flows in a northeastward direction towards the unnamed tributary to Little Mosquito Creek and from there towards Little Mosquito Creek.
- Groundwater in the lower unit of the Columbia aquifer flows in a generally northward direction and does not appear to be influenced by the unnamed tributary.
- The measured vertical hydraulic gradient within the Columbia Group at the FFTA and close to the unnamed tributary to Little Mosquito Creek is downward. This downward vertical gradient indicates that the area is a groundwater recharge area.
- The unnamed tributary is not expected to be a significant groundwater discharge point. The shallowest groundwater (at and near the water table) may discharge into the unnamed tributary. Little Mosquito Creek and its associated wetlands are expected to be the significant groundwater discharge point for the study area.

- A dissolved plume of contamination, primarily defined by the presence of VOCs, exists downgradient of the former fire training pit area.
- The migration of the contaminant plume is essentially confined to the upper flow unit within the Columbia Aquifer. The presence of the silty clay lens at or near sea level appears to control the migration of the contaminants as well as influence the flow within the upper unit.
- No contamination was detected in a monitoring well installed at the projected groundwater discharge point near the confluence of the unnamed tributary and Little Mosquito Creek.
- Subsurface soil samples collected above the water table from the center and within 25 feet of the suspected location of the former fire training pit contain VOC and other contaminants that define the groundwater plume. However, the levels of contaminants currently present in this area are below soil to groundwater screening levels.
- Past operations at the former fire training pit were likely the source of groundwater contamination at the FFTA but the area does not appear to act as a current source of contamination. Past removal actions may have removed the primary source material.
- The detected contaminant levels within the plume have decreased since the 1996 RI sampling.
- The areal extent of the plume appears to be less extensive than it was based on the 1996 RI sampling data.
- Natural attenuation parameters, including dissolved oxygen, oxygen reduction potential and ferrous iron levels, suggest that the magnitude and extent of the plume may be decreasing through natural attenuation and degradation processes.

BASELINE RISK ASSESSMENTS

Baseline human health and ecological risk assessments using the historical and Supplemental RI surface and subsurface soil sample analytical results and Supplemental RI groundwater sample analytical results were performed. The human health risk assessment evaluated potential risks to current industrial workers, future construction workers and hypothetical future residents. The future anticipated land use for the FFTA is the continued use of the area as an airfield. The development of the area for residential purposes is extremely unlikely. This scenario was developed as a baseline for comparison purposes in accordance with EPA guidelines. It should also be noted that for the hypothetical future resident

exposure scenario it was assumed that the shallow FFTA groundwater (Columbia aquifer) would be used for residential purposes. The development of the shallow aquifer as a water supply is highly unlikely in that the lower Yorktown aquifer is more productive and use of the shallow Columbia aquifer is discouraged and restricted by the Commonwealth of Virginia under a groundwater management program administered by the State. Under this program, NASA has abandoned all water supply wells that withdraw from the Columbia aquifer.

The human health risk assessment was performed in accordance with the most recent EPA and VADEQ guidance and in accordance with the procedures and methodologies described in the work plan. Estimated potential risks were calculated for the reasonable maximum exposure (RME) scenario, which represents the conservative, upper end or most reasonably assumed maximum potential exposure to contamination at the FFTA. Carcinogenic and noncarcinogenic risks were calculated. EPA and VADEQ have defined the cancer risk range of 10^{-4} to 10^{-6} as the “target range” or “acceptable risk range” for most sites evaluated under their programs. This risk range represents the potential for contaminant exposure to cause one additional case of cancer in a population of ten-thousand people to one additional case of cancer in a population of one million people. Cumulative cancer risks greater than 10^{-4} generally will indicate that some degree of remediation is required, and cancer risks below 10^{-6} normally will not result in remedial efforts. When the calculated risk falls between 10^{-4} to 10^{-6} , decisions are typically made on a case by case basis. Noncancer risks were evaluated using the Hazard Index (HI) system. EPA and VADEQ have established this system to evaluate the potential for noncarcinogenic health effects of contamination. An HI of 1.0 or less is not expected to result in adverse health effects. The HI for an exposure scenario and for specific target organs are both evaluated under this system. In general, if the total HI exceeds 1.0 then the specific target organs expected to be affected by the contaminants are further evaluated to determine the HI for that organ. The HI value is not a numeric indication of risk. Rather, an HI that exceeds 1.0 indicates that a noncancer health effect cannot be ruled out.

The industrial worker risk scenario represents the current and likely future exposure scenario for the FFTA. This risk scenario assumes an industrial worker will be exposed to surface soils or subsurface and surface soils through the performance of routine maintenance and mowing activities at the FFTA. The cancer risk for exposure to surface soils under this scenario was calculated to be 3.5×10^{-6} . This represents between 3 and 4 additional cases of cancer in a population of one million people which is comparable to the lower end of the EPA acceptable risk range, one case in one million. There were no contaminants detected that would contribute to a noncancer health effect. For potential exposure to subsurface and surface soils, assuming that subsurface soils were disturbed and became available as surface soils, the cancer risk was calculated as 2.8×10^{-6} . This risk is within the acceptable risk range (10^{-4} to 10^{-6}) and represents a risk of about 3 cases in one million industrial workers. There were no contaminants that would contribute to a noncancer health effect.

The construction worker risk scenario was developed to assess the potential risks associated with workers being exposed to surface and subsurface soils and groundwater should the area be developed in the future. The cancer risk for this scenario was calculated to be 4.9×10^{-7} (five in ten million) which is less of a risk than the lower end of acceptable risk range (1×10^{-6} or one in one million). There were no soil contaminants that would contribute to a noncancer health effect. All noncancer groundwater risks were calculated as having an HI of less than 1.0.

The future residential use scenario, although not likely to occur, was developed to evaluate the potential risks associated with residents living at the FFTA and using the shallow groundwater at the site for domestic purposes. Under this scenario it was assumed that subsurface and surface soils would be available for contact and that no soil remediation would be performed and that groundwater would not be treated before use. It was also assumed that a resident would be exposed to the same contaminant levels found at the site from childhood through adult life for a period of 30 years. The cancer risk for this scenario was calculated to be 3.4×10^{-4} (about 3 in ten-thousand) which is a greater risk than the high end of the acceptable risk range, 1×10^{-4} or one in ten-thousand. The potential risk associated with hypothetical use of groundwater was the primary factor in the unacceptable risk. The lifetime cancer risks associated with residential exposure to groundwater and total soils calculated separately were 3.28×10^{-4} and 1.58×10^{-5} , respectively. The potential lifetime cancer risk for soil exposure (one to two in 100,000) is within the acceptable risk range. The primary risk driver in groundwater was arsenic. Other compounds contributing to the risk include pentachlorophenol, benzene, vinyl chloride, and tetrachloroethene. There were no soil contaminants that would contribute to noncancer health effects for exposure to soils under the residential use scenario for both the child and adult receptor. However, an evaluation of the domestic use of site groundwater resulted in HIs greater than 1.0 based primarily on potential exposure to arsenic, iron, manganese, 4-methylphenol, and cis-1,2-dichloroethene in untreated drinking water.

Arsenic was detected in samples from only three of the site monitoring wells. These wells are located within the center of the organic groundwater contamination plume and data from these wells suggest a reducing environment exists within these wells. Samples from two of these wells also contained the only iron concentrations detected above maximum background concentrations and the only detections of 4-methylphenol. In addition, the highest levels of manganese and cis-1,2-dichloroethene were detected in this same area. It is not known if the metals concentrations detected in these wells are directly related to contaminant releases at the site or if the reducing environment related to the degradation of site contaminants has enhanced the dissolution of metals from the geologic materials.

An ecological risk assessment was performed to determine whether adverse ecological impacts are present as a result of exposure to contaminants released to the environment at the FFTA. The habitat, contaminants present, migration pathways, and the routes that receptors may be exposed to

contaminants at the FFTA were defined and evaluated as part of the assessment. The FFTA is a terrestrial habitat and the receptors evaluated were plants, soil invertebrates, and herbivorous and insectivorous bird, mammals and reptiles. In addition, the potential for contaminant exposure through groundwater discharge to surface water and the potential risk to aquatic receptors was evaluated. The contaminant concentrations, occurrence, distribution and potential effects data were evaluated to determine whether adverse effects to growth, survival, and reproduction were likely to occur in these receptors due to exposure to contaminants identified at the FFTA. Overall, risks to plants and invertebrates from chemicals detected at the FFTA in surface soils were found to be low to negligible. Similarly, risks to terrestrial wildlife and aquatic receptors was found to be low and similar to background risks.

ACRONYMS

AOC	Area of Concern
ASTM	American Standards and Text Methods
ATSDR	Agency for Toxic Substances and Disease Registry
AWQC	Ambient Water Quality Criteria
BAF	Bioaccumulation Factor
BERA	Baseline Ecological Risk Assessment
BSAF	Biota Sediment Accumulation Factor
BTAG	Biological Technical Assistance Group
CCME	Canadian Council of Ministers of the Environment
CDI	Chronic Daily Intake
CEC	Cation Exchange Capacity
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CLEAN	Comprehensive Long-Term Environmental Action Navy
COC	Contaminant of Concern
COPC	Contaminant of Potential Concern
CSF	Cancer Slope Factor
CSM	Conceptual Site Model
CTE	Central Tendency Exposure
CTO	Contract Task Order
DCA	Dichloroethane
DCE	Dichloroethene
DNAPL	Dense Non-Aqueous Phase Liquid
DON	Department of Navy
DRO	Diesel Range Organics
EC	Environmental Canada
EEQ	Ecological Effects Quotient
EFANE	Engineering Field Activity North East
EMPCs	Estimated Maximum Possible Concentrations
EPA	Environmental Protection Agency
EPC	Exposure Point Concentration
ERA	Ecological Risk Assessment
FFTA	Former Fire Training Area
FOC	Fractional Organic Carbon
FS	Feasibility Study
FUDS	Formerly Utilized Defense Site
GRO	Gasoline Range Organics

ACRONYMS (Continued)

GSFC	Goddard Space Flight Center
HASP	Health and Safety Plan
HHRA	Human Health Risk Assessment
HI	Hazard Index
HLC	Henry's Law Constant
HQ	Hazard Quotient
I.D.	Inside Diameter
IEUBK	Integrated Exposure and Uptake Biokinetic
IRIS	Integrated Risk Information System
LOAEL	Lowest/Observable/Adverse/Effects/Level
MB	Main Base
ML	Main Land
MNA	Monitored National Attenuation
NOAEL	No Observed Adverse Effects Level
MSL	Mean Sea Level
MTBE	Methyl tert-butyl ether
NACA	National Advisory Committee for Aeronautics
NASA	National Aeronautics and Space Administration
NCEA	National Center for Environmental Assessment
NOAA	National Oceanic and Atmospheric Administration
NOAEL	No/Observable/Adverse/Effects/Level
NTU	Nephelometric Turbidity Units
O.D.	Outside Diameter
ORNL	Oak Ridge National Laboratories
ORP	Oxygen Reduction Potential
OSWER	Office of Solid Waste and Environmental Response
PA	Preliminary Assessment
PAC	Polycyclic Aromatic Compound
PAH	Polynuclear Aromatic Hydrocarbon
PCB	Polychlorinated Biphenyl
PCE	Tetrachloroethene or Perchloroethene
PCDD/PCDF	Polychlorinated Dioxins/Furans
PEF	Particulate Emission Factor
PID	Photoionization Detector
PVC	Polyvinyl Chloride

ACRONYMS (Continued)

QAPP	Quality Assurance Project Plan
RAGS	Risk Assessment Guidance for Superfund
RBC	Risk Based Concentrations
RCRA	Resource Conservation and Recovery Act
RFA	RCRA Assessment Facility
RfD	Reference Dose
RI	Remedial Investigation
RME	Reasonable Maximum Exposure
SAP	Sampling and Analysis Plan
SARA	Superfund Amendment and Reauthorization Act
SERA	Screening Ecological Risk Assessment
SI	Site Inspection
SQG	Soil Quality Guideline
SSL	Soil Screening Level
SVOC	Semi-Volatile Organic Compound
SWCB	State Water Control Board
TAL	Target Analyte List
TCA	Trichloroethane
TCDD	Tetrachlorodibenzodioxin
TCL	Target Compound List
TCLP	Toxic Characteristics Leaching Potential
TEF	Toxicity Equivalency Factor
TEQ	Toxicity Equivalent Concentrations
TOC	Total Organic Carbon
TPH	Total Petroleum Hydrocarbons
TRV	Toxicity Reference Values
UCL	Upper Confidence Level
USACE	United States Army Corps of Engineers
USDA	United States Department of Agriculture
USEPA	United States Environmental Protection Agency
USGS	United States Geologic Society
UST	Underground Storage Tank
UTL	Upper Tolerance Level
VADEQ	Virginia Department of Environmental Quality
VPDES	Virginia Pollution Discharge Elimination System

ACRONYMS (Continued)

VIMS	Virginia Institute of Marine Services
VOC	Volatile Organic Compound
WFF	Wallops Flight Facility
WI	Wallops Island
WWTP	Waste Water Treatment Plant

1.0 INTRODUCTION

This Supplemental Remedial Investigation (RI) has been prepared for the National Aeronautics and Space Administration (NASA) under Contract Task Order (CTO) 848 issued by the Engineering Field Activity Northeast (EFANE) of the Naval Facilities Engineering Command under the Comprehensive Long-Term Environmental Action Navy (CLEAN III) contract number N62467-94-D-0888. The Supplemental RI addresses the Former Fire Training Area (FFTA) Site at the NASA Goddard Space Flight Center (GSFC) Wallops Flight Facility (WFF) located in Accomack County, Virginia. The Supplemental RI activities were conducted and this report prepared as part of the NASA environmental restoration program under the Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA), as amended by the Superfund Amendments and Reauthorization Act (SARA) of 1986, and Virginia Administrative Code Hazardous Waste Management Regulations, 9 VAC20. The Supplemental RI was conducted in accordance with the Final Work Plan for Supplemental RI Activities approved and finalized in January 2003 (TtNUS, 2003a).

1.1 SCOPE AND OBJECTIVES

The primary scope of the Supplemental RI is to perform further investigations and engineering evaluations of the FFTA site to ensure that appropriate remedial alternatives are developed and evaluated. The development and evaluation of remedial alternatives require an understanding of the site characteristics, the nature and extent of contamination, and the identification of potential exposure pathways and receptors. As described in the final work plan, the scope of the Supplemental RI included the performance of additional investigations to fill data gaps and provide supplemental data to refine the risk assessments, to support engineering evaluations, and to provide data needed to develop remedial alternatives for the FFTA site.

The general objectives of the Supplemental RI are to adequately characterize the source area and groundwater conditions at the FFTA, to quantify the potential risks posed to human health and the environment as a result of exposure to those conditions, and to provide sufficient data to evaluate the need and options for remedial actions. The specific objectives include but are not limited to the following:

- Characterization of source area materials.
- Characterization of groundwater conditions.
- Evaluation of the occurrence and/or potential for natural attenuation of contaminants within the aquifer.

- Continued refinement of the Conceptual Site Model (CSM).
- Assessment of potential risks to human health and the environment.

1.2 PLAN ORGANIZATION

This report presents a review of historical and current information and analytical data pertinent to the WFF and the FFTA site and presents a technical evaluation of that data. Section 1.0 provides this introduction and the scope and objectives of the Supplemental RI. Section 2.0 provides a summary of the facility background and environmental investigations. Section 3.0 reviews the FFTA operational and investigation history and CSM, and summarizes the planned Supplemental RI field activities. Section 4.0 presents a discussion of the Supplemental RI activities conducted. Section 5.0 discusses the historical data, presents the results of the Supplemental RI field sampling activities, evaluates the nature and extent of contamination, and refines the CSM. The properties that affect the fate and transport of the contaminants identified at the FFTA are discussed in Section 6.0. Sections 7.0 and 8.0 assess the human and ecological risks associated with potential exposure to contaminants at the FFTA.

2.0 FACILITY BACKGROUND AND HISTORY

2.1 FACILITY DESCRIPTION

2.1.1 Facility Location

WFF is located in Accomack County on the Eastern Shore of the Commonwealth of Virginia (Figure 2-1). The facility is comprised of three separate areas; the Main Base (MB), the Mainland (ML), and Wallops Island (WI). These three areas are in close proximity to each other and total approximately 5,000 acres of landmass and 1,000 acres of marshland. The most heavily developed area is the MB (about 1,900 acres) that includes administrative and technical offices, tracking and data acquisition components, the range control center, rocket motor storage and processing facilities, research and development facilities, an airfield and control tower, aircraft hangars and maintenance facilities, and Navy administration and housing areas. The FFTA is located within the MB area (Figure 2-2).

2.1.2 Facility History

NASA has had a presence in the WFF area since the mid-1940s. From the 1940s until 1959, NASA's predecessor organization, the National Advisory Committee for Aeronautics (NACA), had a limited presence on Wallops Island. In 1959, shortly after its creation, NASA acquired the MB from the Navy. The Navy operated an aviation training field at the MB area from 1942 through 1958. NASA also acquired the ML area at this time and continued to expand land purchases on WI. In 1960, a causeway was constructed from the ML to WI. A detailed history of land ownership and transfers is presented in the Environmental Resources Document, NASA GSFC WFF Wallops Island, Virginia, October 1999, prepared by Occu-Health, Inc. (Occu-Health, 1999).

Beginning in 1942, the Navy operated a training airfield at the MB area, the Chincoteague Naval Auxiliary Air Station, until NASA occupation in 1959. The Navy constructed runways, buildings, and other support facilities throughout the MB area in 1943. The MB facility was used for naval aviation training until 1959 when operations ceased and the property was transferred to NASA. In addition to aviation training, the Navy used the facility for aviation ordnance testing to develop and test aviation ordnance and guided missiles [US Army Corps of Engineers (USACE), 2000].

NASA continues to maintain the runways constructed at the facility by the Navy and occupies many of the structures and buildings that were present at the time of the property transfer. In addition, NASA has expanded and constructed additional buildings within the MB area to support their mission and to provide support to other tenant organizations. The mission of NASA's WFF has undergone several changes

since it was established in 1959, but the main focus has been and continues to be rocket research, the management of suborbital projects, suborbital and orbital tracking, aeronautical research, and space technology research. NASA WFF also provides support to other tenant organizations including the Navy, which maintains surface combat systems and ship systems engineering training centers and uses the airfield and launch sites at WFF. Other tenant organizations include the National Oceanic and Atmospheric Administration (NOAA), which maintains facilities at the MB, the United States Coast Guard, and the Virginia Spaceflight Authority, which utilizes the launch and rocket support facilities (NASA, 2002).

2.2 FACILITY ENVIRONMENTAL INVESTIGATION HISTORY

Formal environmental investigations on a facility-wide basis began at WFF in 1988 with the initiation of an environmental site survey. This survey was conducted for NASA by Ebasco Services, Inc. between 1988 and 1990 and resulted in the identification of 15 areas considered to be potential sites (Metcalf & Eddy, Inc., 1996). Ten of these sites were located at the MB, and the other five were located on WI. A series of initial investigations at these sites culminated in the publishing of a Site Inspection Report in 1992 that was later finalized in 1996 (Metcalf & Eddy, Inc., 1996). The Site Inspection Report detailed the activities and findings for each site. In general, the investigations included the performance of background document reviews, magnetometer studies, soil gas surveys, sampling of soils, groundwater, surface water, and sediments, and the preparation of preliminary Hazard Ranking System scores.

As a result of the site inspection, individual sites were prioritized for additional studies and evaluation. Since 1992, sites have been pursued on individual tracks based on the priority and potential hazard identified. Actions conducted at these sites include supplemental investigations, sampling programs, removals, product recovery, RIs, feasibility studies (FSs), remediation, and closeout. In addition to the formal facility-wide environmental assessment and site inspection process, NASA has conducted other environmental assessments and cleanup programs at areas on both the MB and WI. The Virginia Department of Environmental Quality (VADEQ) has provided monitoring and regulatory support throughout this process. The FFTA has been the subject of removal actions and ongoing investigations including an RI, focused groundwater investigations, and this Supplemental RI.

Environmental investigations at the FFTA began in 1986 after a Commonwealth of Virginia inspection noted the presence of possible petroleum products in soils and in a pit at the fire training area. NASA responded to this finding by conducting a soil excavation and disposal in that same year. From 1989 through 1992, additional investigations including soil gas surveys and soil and groundwater sampling were conducted at the FFTA. Based on the finding that a potential for groundwater contamination existed, NASA initiated RI activities in 1993. RI activities included the completion of soil gas surveys, soil

boring and sampling programs, monitoring well installation and groundwater sampling, and surface soil sampling. Additional groundwater sampling and further human health risk evaluations were completed between 1997 and 2000. Section 3.0 of this report provides a more detailed history of the FFTA, summarizes the analytical findings from previous investigations, and reviews the planned Supplemental RI activities.

2.3 FACILITY ENVIRONMENTAL SETTING

2.3.1 Facility Topography and Drainage

WFF is located on the eastern shore of the Delmarva Peninsula and is relatively flat with no major topographic features. Numerous inlets, tidal estuaries, marshes, bays, and creeks exist throughout the WFF area. The MB is relatively flat with a reported maximum elevation of about 40 feet above mean sea level (msl) (Occu-Health, Inc., 1999). The MB is bordered to the east by extensive marshland and creeks that drain to the Chincoteague Bay and inlet. Little Mosquito Creek, which eventually also flows east into the inlet and to the Atlantic Ocean, borders the MB to the north and west. State Routes 175 and 798 form the southern and southeastern borders of the facility (see Figure 2-2). The runway complex covers a large portion of the MB and forms a flat plateau-like feature that covers the majority of the highest elevations at MB.

Surface water drainage from the MB is through natural and man-made drainage structures. Drainage within the industrialized portions of the MB is controlled and diverted by stormwater collection and conveyance systems. In addition, portions of the base have been isolated from surface water drainage areas by the formation of berms and higher elevation structures. The natural drainage patterns for the southwestern, western, and northern portions of the MB are to Mosquito Creek and its tributaries. The eastern, southeastern, and southern portions of the MB drain to a series of marshlands, creeks, and bays that lead to Chincoteague Bay and the Atlantic Ocean.

Surface waters in the vicinity of the MB are categorized as Class II (Estuarine Waters) according to the Commonwealth of Virginia Water Quality Standards (9 VAC 25-260). The larger, low-lying, receiving waters of the area are tidally influenced and are brackish to saline.

Wetlands are numerous in the WFF vicinity. The open water areas adjacent to and within WFF are mostly bordered by wetlands that generally fall into three categories; marine, estuarine, and palustrine. Marine and estuarine wetlands are tidal wetlands, and palustrine wetlands can be tidal or nontidal. Wetlands in the area of WFF are delineated by the Accomack County Wetlands Board, the Commonwealth of Virginia, and the USACE (Occu-Health, Inc., 1999).

2.3.2 Facility Soils

Soils at WFF are coastal plain soils that are typically nearly level, very deep, and well drained. Soils within the upland, terrace, and level portions of the MB area are predominately Bojac fine sandy loam [US Department of Agriculture (USDA) 1994] (see Figure 2-3). This soil type is reported to be a low organic content, acidic, well-drained soil with moderately rapid permeability. The more steeply sloped areas around the perimeter of the MB and the slopes leading from the level areas to the marshlands are predominantly classified as Molena loamy sand. These soils have properties similar to the Bojac series but reportedly have a higher permeability and are more acidic. Other identified soil types include silty loams and mucky sandy loam (Chincoteague and Polawana series) that occur within the marshland areas around the outer perimeter of the MB.

2.3.3 Area General Hydrogeology

The WFF is located in the Atlantic Coastal Plain Physiographic Province and is underlain by a thick sequence of approximately 7,000 feet of unconsolidated sediments that unconformably overlie crystalline bedrock. The sediments range in age from Recent to Cretaceous and generally consist of alternating series of sands, silts, clays, and gravels. The individual geologic units dip gently to the southeast at a rate of approximately 20 to 80 feet per mile (less than 1 degree) and also thicken in the downdip direction, creating a wedge-shaped section of sediments that gradually thin westward to their updip limit of extent at the Fall Line.

The principal water-bearing units beneath the WFF are, in stratigraphically descending order, the Pleistocene-age Columbia Group and three hydraulically isolated aquifer units within the underlying Miocene-age Yorktown Formation [United States Geological Survey (USGS), 1968]. Collectively, these units form the four major aquifers on the Eastern Shore of Virginia (as identified by the VADEQ) and locally are the primary source of water for public and domestic supplies and for agricultural and industrial uses (Occu-Health, Inc., 1999).

The Columbia Group extends to a subsurface depth of approximately 60 feet and consists of interbedded sands, gravels, and sandy clays deposited under fluvial and marine conditions. The Columbia Group is overlain by a variably thin (generally about 5 feet) veneer of recent deposits composed chiefly of wind-deposited or fluvial sands, silts, and gravels. The groundwater beneath the WFF typically occurs under unconfined conditions within the recent deposits and Columbia Group at depths of 0 to 30 feet (Occu-Health, Inc., 1999). Aquifer tests conducted at the WFF indicate that the hydraulic conductivity of the Columbia Group ranges from about 80 to 200 feet per day, and the specific yield of these deposits ranges from about 0.01 to 0.13 (Occu-Health, Inc., 1999).

The Yorktown Formation is approximately 1,000 feet thick and consists of alternating sequences of fine- to coarse-grained glauconitic sands (that may be variably clayey, silty, or shelly) and finer-grained silts and clays (USGS, 1968; Occu-Health, Inc., 1999). The sands comprise the dominant aquifers, and the silts and clays form aquitards that create confining conditions and that separate the Yorktown Formation into three (upper, middle, and lower) aquifers. At the WFF, the upper Yorktown aquifer generally occurs at a depth of about 100 feet below ground surface and is isolated from the overlying Columbia aquifer by a clay and silt aquitard that is approximately 20 to 40 feet thick. Aquifer tests conducted at the WFF indicated that the transmissivity ranges from about 8,500 to 14,500 gallons per day per foot for the upper aquifer to 4,000 gallons per day per foot for the middle aquifer, and the storativity ranges from about 0.003 for the upper aquifer to about 0.002 to 0.0002 for the middle aquifer.

Aquifer tests conducted at the WFF were reviewed and discussed in the Design Investigation of the Aviation Fuel Tank Farm, or AFTF (Metcalf and Eddy, 1992), to evaluate the degree to which the upper Yorktown aquitard isolates groundwater flow within the Columbia Group and prevents the downward migration of this groundwater into the underlying Yorktown aquifer. This evaluation was performed to satisfy a request by the State Water Control Board (SWCB) for an assessment of the risk of vertical migration of shallow (Columbia Group) groundwater contamination at that site into the underlying Yorktown aquifer due to the pumping of nearby potable supply wells. A review of the design report indicates that there is no single pumping test that can confirm the hypothesis that there is no groundwater migration across the upper aquitard, but, taken as a group in combination with other published materials discussing the regional Yorktown aquifer, the studies suggest that this conclusion is correct.

The aquifer test most pertinent to this current investigation involved the pumping of former supply well D-39 (a shallow well completed in the Columbia Group) for 72 hours, and the measurement of aquifer responses in the Columbia Group and the upper Yorktown Formation in a pair of nearby monitoring wells (Metcalf and Eddy, 1992). The rate of drawdown in the Columbia Group was constant throughout the pumping period, and it was concluded that the constant drawdown (or more specifically, the absence of a reduced rate of drawdown through time) was indicative of a lack of recharge (or leakage) from the underlying Yorktown Formation.

Another test discussed in the Design Investigation (Metcalf and Eddy, 1992) suggested that leakage across the confining unit may occur, but the results from this test were highly qualified and considered inconclusive or questionable. In this test, the pumping of former supply well F-35 (completed in the upper Yorktown Formation) for 24 hours apparently induced drawdown in an adjacent monitoring well completed in the Columbia Group. However, changes in the rates of drawdown in this well were interpreted as potential interferences from the pumping of other nearby supply wells (Metcalf and Eddy, 1992). It was also noted that although the monitoring well used for this test was completed in the Columbia Group, the borehole for

that well was originally drilled into the upper Yorktown Formation. From this, the investigation concluded that if local leakage across the aquitard did occur, it may have been through the breach created by the borehole.

Other aquifer tests presented in the Design Investigation report (Metcalf and Eddy, 1992) were primarily intra-Yorktown tests that demonstrated the efficacy of the Yorktown aquitards in separating the Yorktown into hydraulically isolated upper, middle, and lower aquifers. In one test, the pumping of the middle Yorktown aquifer did not induce a response in the overlying Columbia aquifer. The lack of hydraulic communication between the Middle Yorktown aquifer and the Columbia was attributed to the presences of aquitards within the Yorktown and between the Yorktown and Columbia formations. However, it could not be stated for certain that the presence of the upper aquitard alone would have produced the same effects.

Groundwater flow directions within the Columbia Group throughout the base are variable, but are consistent with the flow directions that are typical or expected for a water-table or unconfined aquifer. That is, the topography of the water table is a generally subdued reflection of the surface topography. The WFF is essentially a peninsula that is surrounded on three sides by surface water bodies that reside at an elevation of sea level and function as discharge points for groundwater, and groundwater generally flows from the facility towards these surface water bodies. As will be discussed in detail in Section 4, the groundwater at the FFTA flows in a generally northeast direction, toward Little Mosquito Creek.

2.3.4 Area Water Supplies

Groundwater is the only source for drinking, agricultural, and process water within the WFF area. This is generally true for a large portion of the Eastern Shore. Because of the reliance on groundwater and the coastal proximity of the area, the Commonwealth of Virginia has declared the Eastern Shore of Virginia a Critical Groundwater Area. The withdrawal of groundwater from the area, including Accomack County, is controlled through the issuance of withdrawal permits under the Virginia Groundwater Management Act. The State Water Control Board, in consultation with local planning commissions and the Department of Health, administers the permitting program and allocates groundwater withdrawal volumes as part of the permitting process.

WFF operates and maintains its own permitted water supply system that includes five wells actively used to withdraw water from the Yorktown Formation. The locations of the WFF MB potable supply wells are illustrated on Figure 2-4. In addition, NOAA and the Town of Chincoteague operate water supply wells located on WFF property. NOAA operates a single well drilled into the Yorktown aquifer. The Town of Chincoteague operates eight supply wells located along the eastern boundary of the WFF MB property. Three of these wells are shallow wells drilled into the Columbia aquifer and the other five wells are deeper

wells installed in the Yorktown aquifer. Table 2.1 presents depth and screened interval information for the supply wells located on the MB facility.

The WFF MB supply wells are not affected by the FFTA, which has locally impacted the Columbia Aquifer. As illustrated on Figure 2-4, the supply wells are all located hydraulically upgradient from the FFTA, as the groundwater originating at the FFTA migrates in a generally northeastward direction and toward Little Mosquito Creek. Most of the supply wells are located at least 4,000 feet upgradient of the FFTA, and because they withdraw water from the Yorktown Aquifer, are also vertically isolated from the FFTA by the intervening upper Yorktown aquitard. The Town of Chincoteague supply wells that withdraw water from the Columbia Aquifer are located approximately 5,000 feet upgradient of the site.

3.0 FORMER FIRE TRAINING AREA (FFTA) BACKGROUND AND HISTORY

3.1 FFTA DESCRIPTION

3.1.1 Facility Location

The FFTA is located on the northern side of the MB adjacent to a former taxiway immediately north of Runway 10-28 (see Figure 3-1). The area is currently an open grass field that gently slopes to the north and northeast. Areas of higher elevation surround the FFTA. There are no surface water bodies in or immediately near the FFTA. Surface runoff within the FFTA flows to low-lying areas within the site where it either infiltrates or evaporates.

The FFTA is bordered to the south by an abandoned taxiway. An earthen berm, about 100 feet long and 4 feet high, exists at the edge of the taxiway. The berm is constructed around a discarded airplane fuselage that was used for fire fighting training exercises. The WFF wastewater treatment plant (WWTP) is located west of the FFTA. To the north, separated by an embankment ranging in height from 3 to 12 feet, is the former Navy magazine area. This area is currently used by NASA as a rocket motor storage and preparation area. Access to this area is controlled and restricted. The area east of the FFTA is heavily wooded with conifer trees and contains a former disposal and debris pile area that was used by the Navy prior to NASA's operations. This area is referred to as the Site 14 Debris Pile and is under investigation by the USACE as part of the federal Formerly Utilized Defense Site (FUDS) program.

3.1.2 FFTA Operational History

NASA began using the FFTA for fire fighting training exercises in 1965 and continued using the area until 1987 (Metcalf & Eddy, Inc., 1996b). It is reported that fire fighting training, conducted twice a week during this time period, consisted of releasing combustible material onto the ground or into an open-top tank, shallow pit, or discarded airplane body, igniting the material, and extinguishing the flames. There are no records identifying the type of materials used during these training exercises.

The open-top tank was reportedly placed on the edge of the taxiway, and an earthen berm was constructed on the downgradient (northern) side of the tank to contain spilled material. The earthen berm still exists and is shown on Figure 3-1. It is reported that a 500-gallon tank was also installed in the area to collect runoff generated during the training exercises. This tank and the open-top burn tank were reportedly removed in 1990 (Metcalf & Eddy, Inc., 1996b).

A fire training burn pit was reportedly located about 100 feet north of the bermed area (Metcalf & Eddy, Inc., 1996b). A review of historical photographs and records conducted as part of the 1996 RI indicated that the pit was unlined and about 2 to 3 feet deep (Metcalf & Eddy, Inc., 1996b). The review did not identify the lateral extent of the pit. The fire training pit and the surrounding soils were reportedly excavated and the area backfilled the area in 1986.

It is not evident from a review of historical documents when use of an airplane fuselage for training purposes began or where that training occurred. It is likely that multiple airplane bodies were used over the years. A airplane fuselage is currently located on the taxiway within the bermed area. In 1994, 7 years after training exercises at the FFTA were terminated, a similar airplane body was located in the field about 300 feet northwest of this area (Metcalf & Eddy, Inc., 1996b).

3.2 FFTA ENVIRONMENTAL INVESTIGATION HISTORY

Environmental investigations began at the FFTA in 1986. In 1986, the Commonwealth of Virginia conducted an inspection of the FFTA and identified substances thought to be jet fuel and crank case oil in the unlined burn pit. The Virginia Department of Waste Management issued a removal order based on the inspection findings. NASA responded to the order by completing a soil removal in November 1986. It is reported that approximately 120 cubic yards of impacted soils were removed. The limits of the excavation were determined by visual observations. No sampling or surveying was conducted as part of the removal action (Metcalf & Eddy, Inc., 1996b).

A preliminary assessment (PA) and site investigation (SI) of the FFTA (Ebasco Services, Inc., 1990) were conducted from 1989 through 1990. The studies included the performance of soil gas surveys, well installation, and surface soil and groundwater sampling. During the initial investigation, 43 soil gas samples were collected from an approximately 1-acre study area consisting of the open field immediately north of the berm and including the suspected location of the training pit. Samples were collected at 100-foot centers and were analyzed using a field organic vapor detection instrument. Samples from three monitoring wells were also collected and analyzed with the field instrument using the headspace technique (Ebasco Services, Inc., 1990). The field data indicated the presence of volatile organic compounds (VOCs) in soil gas and groundwater. The highest levels of soil gas contamination were identified in an area about 80 feet directly north of the berm area. A sample from a monitoring well installed east of the elevated soil gas readings and northeast of the berm area also was found to contain VOCs.

The preliminary field investigation was followed up with an additional sampling program that included the collection of four surface soil samples and additional groundwater samples from the three existing

monitoring wells. The laboratory analytical results indicated that VOCs were not present in the surface soils or an upgradient well, but elevated VOC concentrations were identified in the shallow downgradient well (Metcalf & Eddy, Inc., 1996b).

The analytical data generated from the PA and SI were sufficient to determine the need for additional investigations. However, the exact locations (survey coordinates) of the soil sampling points and the quality of the analytical data were not well defined. For these reasons, the data were used only to guide further investigations but were not considered sufficient to be used for risk assessment or RI purposes.

An RI was conducted at the FFTA from 1993 through 1994. Field activities included a soil gas survey, advancement of soil borings, installation of monitoring wells, and collection of soil (surface and subsurface) and groundwater samples. The study area included the FFTA and the surrounding features and included the field immediately north of the taxiway. The study area extended about 400 feet to the north to the magazine area embankment and about 3,000 feet east-west along the taxiway. The field investigation included sampling in and around other features adjacent to the FFTA including a reported former temporary drum storage area, WWTP sludge pile, and construction debris pile. The drum storage area was not well defined or confirmed during the RI. Its existence and location were described in an earlier account of site conditions, but the drums were removed prior to the RI. Based on previous accounts, the temporary drum storage area was located immediately south of the WWTP sludge pile. These features are shown on Figure 3-1.

Soil gas samples collected from throughout the study area were analyzed using a field laboratory equipped with a gas chromatogram for contaminant identification. The most prevalent compounds identified in this survey were 1,1-dichloroethene (1,1-DCE) and 1,1,1-trichloroethane (1,1,1-TCA). These two compounds were also found at concentrations consistently higher than other compounds. The area approximately 80 feet immediately north of the berm showed the highest concentrations and number of contaminants (Metcalf & Eddy, Inc., 1996b).

Surface soil, subsurface soil, and groundwater samples were analyzed for the full EPA organic Target Compound List (TCL) and inorganic Target Analyte List (TAL) parameters. The groundwater VOC samples were analyzed to achieve low-level detection limits, and samples were analyzed for dissolved metals (filtered samples). Subsurface soils were also analyzed for total petroleum hydrocarbons (TPH), toxicity characteristic leaching potential (TCLP), and engineering parameters. The analytical results underwent full data validation using EPA data validation protocols (Metcalf & Eddy, Inc., 1996b).

Ten surface soil samples were collected as part of the RI field effort. Two of these samples were collected at the WWTP sludge pile and former drum storage area to characterize these adjacent areas.

The remaining eight surface soil samples were collected in the FFTA from around the area that exhibited the elevated soil gas readings. The top several inches of soil were sampled after removing grass and/or debris (Metcalf & Eddy, Inc., 1996b).

Forty-six subsurface soil samples were collected, generally from depths of 5, 10, 15, and 20 feet below ground surface, from 14 boring locations. Eight of the subsurface soil samples were collected from two borings advanced to characterize the WWTP sludge pile. Eighteen subsurface soil samples were collected from five borings advanced in and immediately downgradient of the construction debris piles. Three samples were collected from one boring advanced in an area upgradient of the FFTA. Seventeen subsurface soil samples were collected from six borings advanced in or immediately downgradient of the FFTA (Metcalf & Eddy, Inc., 1996b).

Ten monitoring wells were installed at the site at the conclusion of the soil boring program. Monitoring wells were completed at three depth intervals within the shallow Columbia aquifer. Well locations were selected based on projected groundwater flow direction (to the north and northeast) and proximity to the suspected source area (fire training area). The new and existing monitoring wells were developed using submersible pumps. Groundwater samples were obtained using hand bailers after purging three to five volumes of standing water from each well. However, prior to purging, samples were also obtained from one cross-gradient and three downgradient wells. The stated purpose of these samples was to test for the presence of dense nonaqueous phase liquids (DNAPL). Water level measurements and survey data were collected for all monitoring wells (Metcalf & Eddy, Inc., 1996b).

Based on the RI findings, it was concluded that a groundwater contaminant plume, consisting primarily of benzene, toluene and cis-1,2-dichloroethene (cis-1,2-DCE), was emanating from the former fire training pit area and migrating to the northeast (Metcalf & Eddy, Inc., 1996b). DNAPL was found at the FFTA. The RI findings are presented in more detail in the Supplemental RI Work Plan (TtNUS, 2003a) and discussed in Sections 3.4 and 5.0 of this report.

A supplemental groundwater sampling investigation was conducted in February 2000. The objective of the sampling event was to collect groundwater data to assess if site conditions had changed since the RI data were collected and to update the human health risk assessment (Versar, Inc., 2000). Twelve wells were sampled as part of this effort. One of the existing wells at the site (MW-1) proposed for sampling was not sampled because the well was damaged. All wells were purged of three well volumes prior to sampling using hand bailers or a peristaltic pump. Samples were collected using a hand bailer. All samples were analyzed for hexavalent chromium, VOCs, TPH, semivolatile organic compounds (SVOCs), pesticides, and metals. The analytical data results were validated using EPA protocols (Versar, Inc., 2000).

The results of the 2000 sampling were compared to the RI findings, but no new quantitative risk calculations based on the 2000 data were performed. The report concluded that the apparent groundwater plume had diminished in the source area and broadened in the downgradient area (Versar, Inc., 2000). Section 3.4 of this report discusses the results of the supplemental groundwater sampling in more detail.

3.3 FFTA ENVIRONMENTAL SETTING

3.3.1 Topography and Surface Water

The FFTA is currently an open, grass-covered field that gently slopes to the north and northeast. Several mounded areas of WWTP sludge and construction debris are present at the periphery of the site. The mounds are currently covered with tall grasses and bushes. In general, the site topography is flat with several shallow depressions that collect surface runoff. The elevation of the site is approximately 32 feet above msl along the southern boundary. In the northern portion of the site, surface elevations range from about 27 to 28 feet msl. Along the northern site boundary, the slope increases sharply to a heavily vegetated earthen embankment that separates the FFTA from the former magazine area. To the east, the site area is bordered by low hills and mounds that form the Site 14 Debris Pile area. Surface water runoff within the FFTA accumulates in several low areas within the northern/northeastern portion of the site and then infiltrates or evaporates. During periods of heavy precipitation, surface runoff from the taxiway flows onto the FFTA.

There are no surface water bodies adjacent to the FFTA boundaries that receive surface runoff from the FFTA. The closest water body to the FFTA is an unnamed tributary to Little Mosquito Creek located about 500 feet to the east. The stream originates at the discharge point from the permitted outfall for the stormwater collection network and wash-rack and oil/water separator located within the active airfield south of Runway 10-28 [Virginia Pollution Discharge Elimination System (VPDES) Outfall 003]. The permitted discharge point is shown as the headwaters of the unnamed tributary on Figure 3-1. The unnamed tributary is initially bordered on the eastern and western sides by two FUD sites (Debris Pile 14 and Debris Pile 15) and further downstream flows adjacent to a third FUD site (Abandoned Drum Field) before it discharges into Little Mosquito Creek.

3.3.2 FFTA Soils

The FFTA Site is located within the upland level portion of the MB adjacent to the main east-west runway. The soils at the FFTA belong to the Bojac series as shown on Figure 2-3 (USDA, 1994). These soils are described as well-drained fine sandy loam with low organic content. Because the FFTA is adjacent to a

main runway and considering the past operations at the site, it is likely that the soils at the FFTA have been disturbed and reworked. However, considering the natural topography of the area, it is unlikely that the FFTA area experienced extensive filling with soil materials from outside the immediate area.

3.3.3 Geology as Determined by Previous Investigations

The results of the previous site investigations indicated that the site-specific geology beneath the FFTA is generally consistent with the regional geology as discussed in Section 2.3.3. The Columbia Group is predominantly sandy in nature and consists primarily of light brown to tan silty sand and sand, with traces of clay. A review of site-specific boring logs indicated that clay lenses consisting of highly plastic, fine-grained sandy clays existed within the Columbia Group at an elevation near sea level. These clays appeared to be discontinuous in nature, but the evaluation of their extent was difficult due to the relative incompleteness of the geologic data.

The upper aquitard of the Yorktown Formation was reportedly encountered beneath the FFTA during previous site investigations at a depth of approximately 55 feet, or an elevation of approximately 25 feet below msl (-25 feet). The resultant thickness of the Columbia Group at the FFTA (approximately 55 feet) agrees well with the estimated regional thickness of the unit and supports the identification of the clay at an elevation of -25 feet as the top of the Yorktown Formation. No monitoring wells at the FFTA have been installed within the Yorktown Formation.

3.3.4 Groundwater as Determined by Previous Investigations

Monitoring wells installed during previous investigations have shallow, intermediate, and deep designations. The site conceptual model generated by these investigations however, generally divided the Columbia aquifer into shallow and deep groundwater regimes. Based on the depths of the existing wells, the shallow groundwater regime was interpreted to extend from the water table to a depth of approximately 25 feet below ground surface (approximately sea level), and the deep groundwater regime was interpreted to extend from that point to the top of the Yorktown confining clay. The existing data indicated that the Columbia Group formed a single interconnected aquifer, and its differentiation into shallow and deep zones was not based on hydrogeologic characteristics but rather was to facilitate the delineation of the groundwater plume and the nature and extent of contamination beneath the site.

The depth to groundwater beneath the FFTA was about 14 to 16 feet in January 1994 and February 2000, with the latter elevations generally 0.5 foot higher. Because the groundwater elevation data are limited and both sets of data were acquired during the same season, neither the degree of seasonal fluctuation nor the effects of varying weather cycles (wet versus drought) on the position of the water table were known.

The previous interpretations of groundwater flow conditions indicated that the general direction of flow for the shallow groundwater zone was consistent with and generally mimicked the surface topography, which is common for shallow, unconfined aquifers. Figure 3-2 presents groundwater elevation contours for shallow groundwater as interpreted during the 1996 RI. Groundwater flow direction is interpreted as being to the northeast and following the regional topography (Metcalf & Eddy, Inc., 1996b). The interpretation of flow direction for the deeper groundwater zone beneath the FFTA was constrained by a general lack of data. The few existing data points available during the 1996 RI, as shown in Figure 3-3, indicated that groundwater in the deeper zone generally flowed in a north to northeastward direction (Metcalf & Eddy, Inc., 1996b).

The groundwater elevation contours and interpreted groundwater flow direction generated after a supplemental groundwater sampling event (Versar, Inc., 2000) did not attempt to differentiate two flow regimes. The interpreted groundwater contours presented in that study (see Figure 3-4) indicated that the groundwater in the vicinity of the suspected source area (former fire training pit) generally flowed to the north to northeast and that further downgradient, the groundwater flowed nearly due east toward the unnamed tributary to Little Mosquito Creek.

The previous interpretations therefore, indicated that a significant portion of the groundwater flow in the shallow zone was flowing toward the unnamed tributary. Based on site topography, the wetlands located northeast of the site and adjacent to Little Mosquito Creek, or the creek itself, were interpreted to be the likely discharge points for the majority of the groundwater within the deeper zone of the Columbia aquifer. It could not be determined however, if (similar to the shallow groundwater zone) a component of the deeper groundwater zone also flowed eastward because the wells installed between the source area and the unnamed tributary were all completed at or near the water table, and no hydraulic head data were available for the deeper zone of the Columbia aquifer in this portion of the site.

The data generated during the previous investigations indicated that the hydraulic gradients beneath the FFTA were generally very low but varied depending on location. The horizontal hydraulic gradients within the shallow groundwater zone calculated from data collected in February 2000 (Versar, Inc., 2000) indicated that the horizontal gradient across the site, as calculated between MW-54S and MW-57S (located about 680 feet downgradient), was approximately 0.003. The gradient steepened by an order of magnitude further downgradient, near the discharge point to the unnamed tributary to Little Mosquito Creek, where the horizontal hydraulic gradient calculated between MW-59S (located at the eastern edge of the open field) and WFF14-GW4 (located on the bank of the unnamed tributary to Little Mosquito Creek) was approximately 0.04.

The limited data for the deep groundwater zone indicated that the horizontal hydraulic gradient was generally similar to the shallow zone. The horizontal hydraulic gradient within the deep groundwater zone calculated from data collected in 1996 (Metcalf & Eddy, Inc., 1996b) indicated that the gradient between MW-2D and MW-56D was approximately 0.005. It could not be determined if the gradient steepened in the vicinity of the unnamed tributary (or if the tributary is a discharge point for the deep groundwater), because no deep monitoring wells existed in that area.

The vertical hydraulic gradient throughout the site appeared to be oriented downward, although there were a limited number of well clusters where differences in vertical head were measured at approximately the same horizontal location (for example, the MW-2 cluster). In general though, the deep monitoring wells had lower hydraulic head elevations than the shallow monitoring wells.

3.4 CONCEPTUAL SITE MODEL (CSM)

As described in Section 3.2, a series of environmental investigations and actions occurred at the FFTA from 1986 through 2000. The following sections present an overview of the nature and extent of contamination identified in these investigations and discusses the analytical results and physical characteristics of the FFTA in the form of a general CSM. Many of the earlier investigations did not include the rigorous analytical data handling and validation requirements that are necessary to support the completion of a CERCLA RI or risk assessment. However, that analytical data is referred to for collaborative or comparison purposes where appropriate. The 1996 RI included the collection of surface and subsurface soil samples outside of the FFTA site. The purpose of these samples, as discussed in Section 3.2, was to characterize other features in the general area. The analytical results associated with those samples are not discussed in the following sections.

3.4.1 Surface Water and Sediment

There are no surface water bodies at or in close proximity to the FFTA. As described in Section 3.3.1, surface runoff at the FFTA is contained within the site and either infiltrates or evaporates. Shallow groundwater is projected to discharge to the unnamed tributary to Little Mosquito Creek several thousand feet to the east-northeast of the suspected source area. This tributary originates at a permitted outfall and is part of the stormwater and aviation fueling area oil/water separator discharge management system. No surface water or sediment sampling has been performed at the FFTA.

3.4.2 Soils

Although no soil samples were collected during the performance of the soil gas studies, these studies did identify the possible presence of VOCs in the area of the fire training pit located between MW-54S, MW-55S and MW-2S as depicted in Figure 3-4. The most prevalent contaminants identified in the soil gas survey were 1,1-DCE and 1,1,1-TCA. These two compounds were also found at concentrations consistently higher than other compounds. The area approximately 80 feet immediately north of the berm and 60 feet south of MW-55S showed the highest concentrations and number of contaminants (Metcalf & Eddy, Inc., 1996b).

Eight surface soil samples were collected from around the area that exhibited elevated soil gas readings (generally in the area between MW-54S, MW-55S and MW-2S) during the 1996 RI. All samples were analyzed for TCL organic and TAL inorganic parameters, and the data were validated according to EPA protocols. Metals, low to trace levels of polynuclear hydrocarbons (PAHs), and pesticides were detected in the samples. However, no VOCs were detected in the surface soil samples to substantiate the elevated soil gas readings (Metcalf & Eddy, Inc., 1996b). The analytical results for these surface soil samples are presented in Section 5.0 of this report.

Seventeen subsurface soil samples were collected from six borings advanced at the FFTA site during the 1996 RI. The soil borings were placed in areas immediately upgradient and downgradient of the suspected source area (near MW-54S and MW-55S) and within the suspected downgradient contaminant migration area (MW-58S, MW-61I, MW-56D and MW-57S) based on projected groundwater flow patterns.

All subsurface soil samples were analyzed for TCL organic and TAL inorganic parameters, and the data were validated according to EPA protocols. Metals were detected in all subsurface soil samples but at levels that were consistent throughout the site. Trace levels of pesticides and SVOCs were detected in isolated samples. A sample collected from the 15-foot depth in the boring located near MW-55S (SB-8), immediately downgradient of the elevated soil gas area, contained the highest level of SVOCs. This sample was described as stained and exhibiting a strong petroleum-like odor (Metcalf & Eddy, Inc., 1996b). In addition, this sample was the only subsurface soil sample that contained VOCs. Toluene (26 ug/kg), xylene (27 ug/kg) and estimated levels of 1,1,1-trichloroethane (1,1,1-TCA; 3 ug/kg), cis-1,2-DCE (3 ug/kg) and ethylbenzene (10 ug/kg) were detected in this sample. No subsurface soil samples were collected from the area that exhibited the highest levels of soil gas contamination or within the area identified as the suspected location of the former fire training pit. The analytical results for the subsurface soil samples are presented in Section 5.0 of this report.

3.4.3 Groundwater

Thirteen monitoring wells at the FFTA site and an off-site background well (MW-34S) were sampled during the 1996 RI. All samples were analyzed for VOCs, SVOCs, polychlorinated biphenyls (PCBs), pesticides, and metals (total and dissolved). Twelve of the FFTA wells (MW-1 was damaged and was not sampled) were resampled in 2000. The 2000 supplemental groundwater samples were analyzed for VOCs, SVOCs, metals, pesticides, and general chemistry parameters. The data were subsequently validated by an independent, third-party validation subcontractor in accordance with EPA Region 3 guidelines and procedures (Metcalf & Eddy, Inc., 1996b; Versar, Inc., 2000).

Analytical results from the 1996 RI indicated significant detections of VOCs, a significant detection of SVOCs, the presence of metals, and trace isolated detections of pesticides. The analytical results were evaluated in the 1996 RI to identify the significance and nature and extent of the groundwater contamination. This evaluation included a comparison of the groundwater contaminant levels to EPA Region 3 risk-based concentrations (RBCs) and an examination of the occurrence and distribution of the detected contaminants. The evaluation considered analytical results from samples collected before and after purging of the monitoring wells (Metcalf & Eddy, Inc., 1996b). As a result of this evaluation, benzene, toluene and cis-1,2-DCE were identified as the predominant contaminants that characterized the extent and significance of the contamination.

Benzene was detected at concentrations ranging from an estimated concentration of 9 ug/L to 120 ug/L in three monitoring wells. Based on these detections, a benzene plume was estimated as shown in Figure 3-5. The projected plume extends from the general former fire training area downgradient in a northeasterly direction (Metcalf & Eddy, Inc., 1996b). It was noted however, that the downgradient detections of benzene were in samples collected before purging and that benzene was not detected in any of the samples collected from downgradient wells after purging.

Toluene was detected in two wells immediately downgradient of the suspected former fire training pit at 32 ug/L and 1,800 ug/L. A toluene plume, centered on monitoring wells MW-55S and MW-55D, was projected based on these two samples as shown in Figure 3-6 (Metcalf & Eddy, Inc., 1996b).

Cis-1,2-DCE was detected in eight of the monitoring wells at concentrations ranging from 1 ug/L to 3,000 ug/L (including analytical results from samples collected before and after purging). A cis-1,2 DCE plume (using the highest detected concentrations detected in pre- and post-purging sample results) was projected as extending from the former fire training area in a downgradient (northeastern direction, centered on monitoring well MW-611 (see Figure 3-7) (Metcalf & Eddy, Inc., 1996b).

Significant detections of SVOCs including naphthalene (2,000 ug/L), 2-methylnaphthalene (3,000 ug/L), and phenanthrene (82 ug/L) were measured in the sample collected from monitoring well MW-55S. The highest levels and greatest number of metals, considering filtered and unfiltered samples, were detected in MW-54S and MW55S. The filtered (dissolved) metals concentrations were highest in the sample collected from MW-55S (Metcalf & Eddy, Inc., 1996b). This monitoring well is located immediately downgradient of the suspected former fire training pit.

The analytical results from the supplemental groundwater sampling study conducted in 2000 indicated a similar contaminant pattern. Benzene was detected in the same two downgradient wells (MW-61I and MW-56D) as in 1996 but at lower concentrations (maximum of 31 ug/L). Based on these two detections, a smaller benzene plume, centered on MW-61I, was projected in the downgradient (northeastern) portion of the FFTA site (see Figure 3-8) (Versar, Inc., 2000). Toluene was detected at concentrations slightly lower than in 1996 (maximum 1,300 ug/L) but in the same wells and in a downgradient well. Figure 3-9 presents the toluene plume projected using the 2000 analytical results (Versar, Inc., 2000). The 2000 sampling results identified slightly lower levels of cis-1,2-DCE (maximum concentration 1,700 ug/L) and projected a similar plume centered around MW61I but, based on contaminant patterns and groundwater contours, extending to the east, see Figure 3-10 (Versar, Inc., 2000).

The 1996 RI and the 2000 groundwater sampling event did not, because of the lack of available deep monitoring wells, evaluate the vertical extent of the contamination. Similarly, (because of the lack of deeper monitoring wells), the potential impact of the deeper flow zone, interpreted to exist within the Columbia Group in the 1996 RI, on contaminant migration was not evaluated.

The semivolatile and dissolved metals analytical results from the 2000 sampling were similar to the 1996 results. Monitoring well MW-55S was the only well that indicated the presence of any significant concentrations semivolatile compounds (naphthalene and phenanthrene). Dissolved metals were detected at the greatest concentrations in wells MW-54S and MW-55S.

3.4.4.4 Summary of CSM as Determined by Previous Investigations

Based on the results of the 1996 RI (soil gas, surface soil, subsurface soil, groundwater) and 2000 (groundwater only) sampling events, past fire fighting training exercises consisting of the releasing and igniting of combustible materials were identified as the source of contamination north of the abandoned taxiway adjacent to Runway 10-28. Based on the existing analytical data and historical accounts of training exercises, the source area of contamination was located between the taxiway, MW-55S, MW-54S, and MW-02S. It was concluded that groundwater in the area occurs in an unconfined, water table aquifer within the Columbia Group that is isolated by an aquitard from the lower Yorktown Formation. Groundwater was

projected to flow in a northeasterly direction, and existing analytical data suggested that contamination was migrating with the groundwater flow. The primary contaminants identified in the downgradient area consisted of VOCs.

This CSM is limited by the available site-specific data. Although a removal action in the suspected source area (former fire training pit) was conducted in 1986, no soil samples were collected at that time and no subsurface investigations have been conducted since. A series of monitoring wells have been installed upgradient and downgradient of the suspected source area but not within the area exhibiting the highest soil gas readings. The monitoring wells are screened at varying depths within the Columbia Group to monitor shallow and deeper portions of the aquifer. However, shallow and deep monitoring wells have not been constructed in pairs, and few deeper monitoring wells in the projected downgradient direction exist in the study area. Because of the lack of deeper monitoring wells, the possible variation in the nature and extent of the contamination with depth has not been evaluated. The existing groundwater data do not include sufficient parameters to evaluate the degradation or natural attenuation processes that may be occurring within the contamination zone that may impact the evaluation of potential remedial alternatives for the site.

3.5 PLANNED SUPPLEMENTAL REMEDIAL INVESTIGATION

Supplemental RI activities to refine the CSM and meet the objectives identified in Section 1.1 of this report were defined in a final work plan (TtNUS, 2003a). That work plan includes a detailed description of the field effort, data handling and management, and data evaluation process and methodologies to be applied in completing this Supplemental RI. The work plan includes a Sampling and Analysis Plan (SAP) and a Quality Assurance Project Plan (QAPP) that provide a discussion of methods and procedures to be applied in performing the Supplemental RI. The sampling program presented in the work plan is summarized in the following sections. Pesticide analysis was not included in any of the Supplemental RI samples. This decision was based on a review of existing historical information and analytical data, which indicates that pesticides were not associated with any site-specific activity. In addition, because of the lack of an association with any site-specific activity, historical pesticide results were not to be included in any supplemental data evaluations.

3.5.1 Background Study

The work plan presents a review of background surface and subsurface soil and groundwater data available from historical investigations conducted at WFF. In addition, the work plan discusses the completion of a separate study to be conducted within the MB to characterize the two major soil types (Bojac and Molena) present within the MB and the collection of additional groundwater data. The completion of the MB background study includes the evaluation of existing data and the compilation of historical and new data for

application under this Supplemental RI. No background soil samples were planned as part of the Supplemental RI program. Site-specific background groundwater samples were to be collected from monitoring wells MW-1, MW-53S, 14-GW1, and 14-GW2. These samples were to be analyzed for VOCs plus Methyl tert-butyl ether (MTBE), SVOCs, PCBs, TAL metals, TAL dissolved metals (field-filtered), and monitored natural attenuation (MNA) parameters (hydrogen, soluble manganese, nitrate, chloride, sulfate, methane/ethane/ethene, and TOC). Groundwater field parameters (pH, temperature, dissolved oxygen, oxygen reduction potential (ORP), salinity, conductivity, turbidity, alkalinity, ferrous iron, and hydrogen sulfide) will also to be measured at each well. Additionally, monitoring well 53S was to be analyzed for perchlorate.

3.5.2 Soils

Soil sampling activities to be conducted to further delineate the nature and extent of contaminated soil in the vicinity of the suspected source area (former burn pit) were to consist of collecting 18 surface (0 to 6 inches in depth) and 36 subsurface soil samples from 18 different locations. The 18 sampling locations were to be located in concentric circles centered in the suspected source area.

A surface soil sample was to be collected from each location. In addition, a boring was to be advanced to the water table at each location to characterize the subsurface materials and allow for the collection of subsurface samples. Two subsurface soil samples were to be obtained from each boring. The first sample was to be collected from a depth of 2 to 4 feet. The second sample was to be collected at a depth determined in the field based on visual observations, monitoring instrument readings, and water table depth. If staining or elevated instrument readings were encountered, the sample was to be collected at that depth. If neither of these occurred, the second sample was to be collected at halfway between the ground surface and the water table.

Samples were to be analyzed for VOCs (including MTBE), SVOCs, PCBs, and TAL metals. Surface soil samples were also to be analyzed for total organic carbon (TOC), pH, and grain size. Additionally, surface soil samples and any subsurface soil exhibiting waste or charred characteristics were to be analyzed for dioxin (and furans).

3.5.3 Groundwater

The groundwater investigation was to consist of the installation of new monitoring wells, an evaluation of existing wells and their redevelopment (if necessary), the performance of two synoptic rounds of groundwater elevation measurements, and the collection of groundwater samples. Six new monitoring wells were to be installed at the FFTA. The purpose of the new wells was to fill data gaps by providing a more

complete delineation of groundwater flow patterns (by providing additional hydraulic head data) and a more complete delineation of the nature and extent of groundwater contamination.

Three shallow wells screened across the water table and three deep wells, screened near the base of the Columbia Group were to be installed. One shallow well was to be installed in the center of the suspected source area. One shallow and deep well pair was to be installed to the east toward the unnamed tributary to Little Mosquito Creek. One shallow well was to be installed in the far northeastern direction downgradient of the site, near the mouth of the unnamed tributary. Two deep wells, one located in the northeast portion of the study area near monitoring well MW-57S, and one in the north central portion of the study area near monitoring well MW-58S were to be installed.

Groundwater samples were to be obtained from the new and existing wells after the wells were developed. The groundwater samples were to be obtained via low-flow sampling techniques following appropriate EPA Region 3 protocol and guidelines and analyzed for VOCs plus MTBE, SVOCs, PCBs, TAL metals, TAL dissolved metals (field-filtered), and MNA parameters (hydrogen, soluble manganese, nitrate, chloride, sulfate, methane/ethane/ethene, and TOC). Groundwater field parameters (pH, temperature, dissolved oxygen, ORP, salinity, conductivity, turbidity, alkalinity, ferrous iron, and hydrogen sulfide) were to be measured at each well. Additionally monitoring wells MW-2S, MW-55S, MW-61I, and MW-56D, were to be analyzed for perchlorate. Perchlorate analysis was included for these wells to collect data as part of a facility-wide sampling effort to assess the presence or absence of perchlorate. There is no information to suggest that the waste handled at the FFTA contained perchlorate or that materials such as missile propellants or motors were associated with any activity at the site.

4.0 SUPPLEMENTAL RI INVESTIGATIONS

4.1 BACKGROUND STUDY

A background study of soil and groundwater for the WFF MB area was conducted in 2003. The results of that study are presented in a report, Background Soils and Groundwater Report for the Main Base at National Aeronautics and Space Administration (NASA) Wallops Flight Facility (WFF), draft dated May 2003 (TtNUS, 2003b). The background study included the evaluation of existing background samples as well as the collection and analysis of additional soil and groundwater samples from locations identified as being outside of the influence of known or suspected waste sites or sources of contamination. A separate work plan for the background study, identifying the location of sampling points, analytical protocols, data management, and data evaluation methodologies was approved by the EPA and VADEQ.

The background study included a comparison and statistical evaluation of the similarities and differences among chemical/analyte concentrations detected in the two major soil series (Bojac and Molena) present at the MB. In addition, the study evaluated the similarities and differences between chemical/analyte concentrations detected in surface and subsurface samples within the same soil series. The background study concluded that there are sufficient differences in metals concentrations between Bojac and Molena series soils to consider them dissimilar for background data comparisons. In addition, the study concluded, based on the occurrence and distribution of concentrations of a number of metals, that the surface and subsurface profiles for each soil series were dissimilar and suggested that a compositional difference exists with depth (TtNUS, 2003b). For organics, the study concluded that surface and subsurface soils were dissimilar but that no significant differences were apparent between the two soil series. The background study presents the evaluation of the background soils in detail and includes tables of analytical results for each soil series and depth (surface and subsurface).

The soils at the FFTA have been categorized as belonging to the Bojac series. As indicated above, an evaluation of surface and subsurface background samples within the Bojac series indicates that surface and subsurface soils should be evaluated separately. For the purposes of this Supplemental RI, the background data set established in the referenced background report (TtNUS, 2003b) will be used to evaluate site-specific soil samples collected and analyzed from the FFTA. Tables 4.1 through 4.4 present the occurrence and distribution of substances detected in Bojac surface and subsurface soil samples, respectively. Figures 4-1 and 4-2 show the background soil sample locations.

As indicated above, the background study included the evaluation, collection, and analysis of background groundwater samples. The wells identified as FFTA site-specific wells, MW-1, MW-53S, 14-GW1, and 14-GW2, were sampled during the background study and during the Supplemental RI. The analytical results

for both sampling events are shown in Appendix A. Figure 4-2 shows the location of the MB background monitoring wells.

4.2 SOIL INVESTIGATION AND SAMPLING

Section 3.5 presents the planned Supplemental RI activities as proposed in the work plan (TtNUS, 2003a). Unless indicated below, all work was performed in accordance with the work plan. Deviations from the work plan are detailed, and a summary of field activities conducted is included in the following sections. Significant site conditions observed during these activities are also presented. All sampling locations were surveyed by a licensed professional surveyor, and locations were tied back into benchmarks referenced to the Virginia State Plane Coordinate System, South Zone, North American Datum 1983 (NAD 1983) High Accuracy Recovery Network (HARN) Adjustment, 1993. Elevations were referenced to the National Geodetic Vertical Datum 1929 (NGVD 29). Survey data are presented Appendix B.

4.2.1 Surface Soil

A total of 18 surface soil samples (0 to 6 inches) plus quality assurance/quality control (QA/QC) samples (e.g., duplicates, blanks) were proposed in the work plan for analysis for VOCs, SVOCs, PCBs, TAL metals, dioxin (and furans), pH, TOC, and grain size.

4.2.1.1 Deviations From the Work Plan

There were no deviations from the sampling plan for surface soil samples.

4.2.1.2 Surface Soil Sampling and Analysis

Surface soil sampling was conducted from February 3, through February 5, 2003. A total of 18 sampling locations were field-marked in the study area along sampling vectors radiating from the center of the suspected source area (former fire training pit). Sample location FTA-SS-101 was located in this area. Other locations were placed at the points where vectors crossed distance rings concentrically drawn around the source area. The sampling interval along each vector was 25 feet for the first sampling ring around the center point and 50 feet for the additional rings; sampling locations were adjusted based on field conditions. A surface soil sample consisting of soils from 0 to 6 inches was collected from each location and analyzed for VOCs, SVOCs, PCBs, TAL metals, dioxin (and furans), pH, TOC, and grain size. Figure 4-3 shows the Supplemental RI surface soil sampling locations. Sample logs are included in Appendix C.

4.2.2 Subsurface Soil

As detailed in Section 3.5, soil borings were to be advanced at each surface soil location with two subsurface soil samples proposed from each boring for a total of 36 subsurface soil samples. Planned analysis was for VOCs, SVOCs, PCBs, and TAL metals.

4.2.2.1 Deviations from the Soil Sampling Work Plan

An additional soil sample was collected from boring FTA-101 at a depth of 16 to 17 feet (sample FTA-SU-101-1617) and analyzed for the following: VOCs, SVOCs, PCBs, TAL metals and dioxin. This was an "opportunity sample" collected because a black material described as "charred or charcoal-like" was observed in the sampling sleeve. This sample was located just below the soil/water interface.

4.2.2.2 Subsurface Soil Sampling and Analysis

From February 3, through February 5, 2003, borings were advanced to the water table at each of the 18 locations to characterize the subsurface materials and allow for the collection of subsurface samples. Borings were advanced using the Geoprobe direct push method, and the subsurface materials were retrieved using a macrocore sampler with the recovered material collected in an acetate sleeve. A 2-inch-diameter macrocore sampler was used to allow for sufficient volume to log the subsurface lithologies and to collect analytical samples. Two subsurface soil samples were obtained from each boring location. The first sample was collected from a depth of 2 to 4 feet. The second sample was collected at a depth determined by the field scientist based on visual observations and with the aid of a photoionization detector (PID). If visual staining of the soils was apparent, the second sample was collected from the depth that staining was observed. If no staining was evident, the depth exhibiting the highest PID reading was sampled. If neither visual staining nor elevated PID readings were encountered in a boring, a sample was collected from halfway between the ground surface and the water table (as determined by the static water elevation measured in monitoring well MW-55S on the day of the soil sampling). As detailed above, an additional sample was collected from FTA-SB-101 at a depth of 16 to 17 feet due to the presence of charred material in the sample. Therefore, a total of 37 subsurface samples were collected in addition to QA/QC samples. Samples were analyzed for VOCs (including MTBE), SVOCs, PCBs, and TAL metals. In addition, sample FTA-SU-1617 was also sampled for dioxin (and furans). Figures 4-4 and 4-5 show soil boring locations and subsurface soil sampling locations, respectively. Sample logs and boring logs are included in Appendices C and D, respectively.

Several soil borings exhibited elevated PID readings and fuel or solvent odors. These locations included FTA-SB-101, FTA-SB-104, FTA-SB-105, FTA-SB-106, FTA-SB-107, FTA-SB-108, FTA-SB-109 and FTA-

SB-110. None of the borings had visual evidence of contamination with the exception of the bottom interval at FTA-SB-101 which contained a 0.5 feet thick layer of black "charred or charcoal-like" material at the water table (16 to 17 feet in depth). This location was sampled as an "opportunity sample" as discussed previously. The boring logs provide details of lithology, PID readings, and other visual and olfactory observations. A summary of each boring follows:

- FTA-SB-101: Sand with silt with a solvent odor and PID readings observed at a depth of 7 feet and charred material at soil/water interface at 16 to 17 feet. PID readings ranged from 99 to 312 ppm, with the highest readings at 16 to 17 feet. A fuel oil-like product was also observed on soils from the bottom interval. The boring was terminated at 17 feet.
- FTA-SB-102: Sand with silt. Boring terminated at 16 feet. No unusual odors or PID readings.
- FTA-SB-103: Sand with trace amount of silt. Boring terminated at 20 feet. No unusual odors or PID readings.
- FTA-SB-104: Sand with silt with a fuel or solvent odor and PID readings encountered beginning at a depth of 5 to 6 feet. PID readings ranged from 20 to 90 ppm, with highest readings from 8 to 10 feet. Boring terminated at 16 feet.
- FTA-SB-105: Sand with a trace of silt. Boring terminated at 16 feet. Elevated PID readings and fuel odor observed starting at a depth of 3 feet. Odors became stronger at 9 feet. PID readings ranged from 9 to 209 ppm with the lowest levels at 3 feet and the highest levels at the bottom of the boring.
- FTA-SB-106: Sand with some silt. Boring terminated at 16 feet. PID readings and slight fuel odor noticed at 12 to 16 feet, with PID levels ranging from 10 to 29 ppm.
- FTA-SB-107: Sand with some silt. Boring terminated at 16 feet. PID readings and slight fuel odor noticed at 12 feet with increasing PID readings and stronger odors to 16 feet. PID readings ranging from 10 to 93 ppm.
- FTA-SB-108: Sand with silt. No odors or PID readings until bottom interval of 15 to 16 feet where a slight fuel odor was noticed and a PID reading of 3 ppm was recorded.
- FTA-SB-109: Sand with silt. No odors or PID readings until bottom interval of 15 to 16 feet where a slight fuel odor was noticed and a PID reading of 4 ppm was recorded.

- FTA-SB-110: Sand with silt. No odors or PID readings until bottom interval of 15 to 16 feet where a fuel odor was noticed and PID readings up to 120 ppm were recorded.
- FTA-SB-111: Sand with trace amount of silt. Boring terminated at 20 feet. No unusual odors or PID readings.
- FTA-SB-112: Sand with trace amount of silt. Boring terminated at 20 feet. No unusual odors or PID readings.
- FTA-SB-113: Sand with trace amount of silt. Boring terminated at 16 feet. No unusual odors or PID readings.
- FTA-SB-114: Sand with trace amount of silt. Boring terminated at 20 feet. No unusual odors or PID readings.
- FTA-SB-115: Sand with trace amount of silt. Boring terminated at 20 feet. No unusual odors or PID readings.
- FTA-SB-116: Sand with trace amount of silt. Boring terminated at 16 feet. No unusual odors or PID readings.
- FTA-SB-117: Sand with trace amount of silt. Boring terminated at 16 feet. No unusual odors or PID readings.
- FTA-SB-118: Sand with trace amount of silt. Boring terminated at 16 feet. No unusual odors or PID readings.

4.3 HYDROGEOLOGIC INVESTIGATION

As detailed in Section 3.5.3, the groundwater investigation work plan called for the installation of six new monitoring wells, collection of two rounds of water level measurements, and collection and analysis of groundwater samples. All new and existing monitoring wells were surveyed by a licensed professional surveyor and locations were tied back into benchmarks referenced to the Virginia State Plane Coordinate System, South Zone, North American Datum 1983 (NAD 1983) High Accuracy Recovery Network (HARN) Adjustment, 1993. Elevations were referenced to the National Geodetic Vertical Datum 1929 (NGVD 29). Survey data are presented in Appendix B. Field activities performed are detailed below.

4.3.1 Monitoring Well Installation

The work plan called for installation of three shallow wells screened across the water table and three deep wells screened near the base of the Columbia Group. Rationale for well placement is contained in the work plan and Section 3.5.3 of this report.

4.3.1.1 Deviations from the Work Plan

Drilling, construction, and development of new monitoring wells was performed in accordance with the work plan with the following exceptions:

- During the well drilling operations, the lithology from the surface to the ground water was not logged for MW-101 because this information was previously obtained for soil boring FTA-SB-101.
- The boring at MW-103D was continuously logged during drilling. Due to the presence of a clay layer and water at 20 to 25 feet, it was determined that an additional well (MW-103I) would be constructed. MW-103S and MW-103I were constructed with 5-foot screens rather than the 10-foot screens specified in the work plan. MW-103S was screened from 10.5 to 15.5 feet below ground surface and MW-103I was screened from 20 to 25 feet below ground surface. MW-103D was screened below the clay layer; and a 10-foot screen was used for construction. The screened interval was 38 to 48 feet below ground surface.
- The work plan indicated that monitoring well development would be completed with a submersible pump. However, due to low yields at WFF14-03, WFF14-04 and FTA-MW-103S, a peristaltic pump and disposable bailer were used to develop the wells.

4.3.1.2 Monitoring Well Drilling, Installation, and Development

From January 28, through February 25, 2003, well drilling, installation, and development activities were performed. Seven new monitoring wells were installed at the FFTA; the locations of the new (and existing) wells are illustrated in Figure 4-6. The purpose of the new wells was to fill data gaps by providing a more complete delineation of groundwater flow patterns (by providing additional hydraulic head data) and a more complete delineation of the nature and extent of groundwater contamination.

The screened intervals for the new wells varied depending on the particular data needs at that location (see Table 4-5). Four shallow wells screened across the water table and three deep wells designed to monitor the lower portion of the Columbia aquifer were installed as follows:

- Shallow well FTA-MW-101S was installed near the center of the former fire training pit in the area of high soil gas readings as measured during previous investigations. This well was installed to investigate the quality of the groundwater at the source area.
- Shallow well FTA-MW-103S, intermediate well FTA-MW-103I, and deep well FTA-MW-103D well were installed in the area between the FFTA and the unnamed tributary to Little Mosquito Creek. The purpose of this cluster was to monitor the quality of the shallow groundwater that discharges to the unnamed tributary and to provide hydraulic head and chemical data needed to determine if any portion of the groundwater plume within the lower portion of the Columbia aquifer is migrating from the FFTA to the unnamed tributary.
- Shallow well FTA-MW-104S well was installed near the wetlands bordering Little Mosquito Creek and its unnamed tributary to investigate the quality of the groundwater at the hypothesized and conceptual discharge point for the FFTA groundwater.
- Deep well FTA-MW-105D was installed in the farthest downgradient (northeast) corner of the FFTA accessible to a drilling rig to determine whether part (or all) of the groundwater plume (especially the portion of the plume within the lower portion of the Columbia aquifer) migrates northeastward or directly toward Little Mosquito Creek rather than eastward and toward the unnamed tributary.
- Deep well FTA-MW-102D was installed near existing shallow well MW58S to determine whether there is a significant northerly component of groundwater flow within the lower portion of the Columbia aquifer.

The wells were installed using the hollow-stem auger drilling method. An all-terrain rig was required to install MW-104S due to the steep terrain. Continuous split-spoon samples were obtained and lithologically logged for each borehole. For each shallow well, the borehole was advanced to a depth of approximately 4 feet below the water table, and a 10-foot length of screen (except where indicated in the prior section) was installed within the borehole. For the deep wells, the borehole was advanced to the top of the Yorktown confining clay (approximately 55 feet deep), and a 10-foot length screen was installed within the borehole.

The wells were constructed with 2-inch-diameter PVC well screens and risers. The well screen openings were 0.010 inch wide (10-slot). The filter pack consisted of appropriately sized sand (Morie No. 1 or equivalent) and extended from the bottom of the borehole to a height of approximately 2 feet above the top of the screen. The filter pack was emplaced by pouring the sand into the borehole and around the screen. An annular seal consisting of bentonite chips or pellets was installed above the filter pack. The seal was a minimum thickness of 2 feet and was allowed to fully hydrate per the manufacturer's recommendations.

before the remainder of the borehole was backfilled. The bentonite was activated by pouring clean, potable water into the borehole. The remainder of the annular space was backfilled with a bentonite/cement grout (approximately 5 pounds of bentonite and 7 gallons of potable water per 94-pound bag of Portland cement). A concrete collar was installed from a depth of 2 feet to the ground surface and around the steel protective standpipe.

Significant observations recorded during well installation activities include:

- Running sands were encountered during the installation of the deep wells at locations FTA-MW-102D, FTA-MW-103D and FTA-MW-105D at 38 to 40 feet, 40 feet, and 42 feet, respectively. The total boring depths on these wells were 48, 50, and 52 feet, respectively. The wells were installed using a center plug in the auger bit and potable water to wash out the accumulated sands before the center plug was knocked out and the screen and casing installed.
- When installing the well at FTA-MW-103I, bentonite pellets bridged at 15 feet and the drillers pulled out the remaining auger flights to see if the borehole would remain open to continue well construction. The borehole collapsed and a new boring was completed approximately 5 feet south of the original location.
- Flowing sand was encountered at well location FTA-MW-104S at 11.5 feet. The borehole collapsed and had to be abandoned. The hole was backfilled with bentonite pellets and a new borehole drilled.

All existing monitoring wells were redeveloped, and all new monitoring wells were developed. Wells were developed with a submersible pump, except where low yields required the use of a peristaltic pump. A surge block was used as an aid in well development. Turbidity was monitored during development, and well development continued until a turbidity value of less than 10 NTU was achieved, or for a maximum of 3 hours. The field geologist recorded the field-measured parameters of pH, conductivity, dissolved oxygen, and turbidity on well development logs. Groundwater levels were measured prior to and after well development activities. All groundwater generated during well development activities was containerized for subsequent characterization and appropriate disposal.

Significant observations recorded during well development are presented below:

- Elevated PID readings were recorded during the development of monitoring wells FTA-MW-55S, FTA-MW-55D, FTA-MW-56D, FTA-MW-57S, FTA-MW-58S, FTA-MW-61I, and MW-101S.

- FTA-MW-55D had a slight sheen on the water at the beginning of development. At the end of development there was no sheen observed on the surface of the discharge tank. A petroleum odor was also noticed during development.
- FTA-MW-55S had a petroleum odor during development.
- Sulfur odors were noted during the development of FTA-MW-57S and FTA-MW-611.
- An organic odor was noted during development of FTA-MW-58S.

4.3.2 Water Level Measurements

Two rounds of synoptic water level measurements were conducted. The first round, prior to monitoring well sampling, was performed on March 3, 2003 after a significant snowfall/melt event. The second round of water level measurements was performed on March 19, 2003, several days after the conclusion of monitoring well sampling. Water level measurements and well construction and elevation data are presented in Table 4-5.

4.3.3 Monitoring Well Sampling and Analysis

As detailed in the work plan, a complete round of groundwater sampling was to be conducted a minimum of 10 days after the development of the monitoring wells. Groundwater samples were to be collected for analysis for VOCs, SVOCs, PCBs, TAL metals, TAL dissolved metals, and MNA parameters. Groundwater field parameters were also measured and recorded. Four wells were also scheduled for perchlorate analysis.

4.3.3.1 Deviations from the Work Plan

There were no deviations from the work plan with regard to monitoring well sampling and analysis.

4.3.3.2 Monitoring Well Sampling

Monitoring well sampling was conducted in accordance with the work plan from March 3, through March 14, 2003. The groundwater samples were obtained via low-flow sampling techniques following appropriate EPA Region 3 protocol and guidelines, and analyzed for VOCs plus MTBE, SVOCs, PCBs, TAL metals, TAL dissolved metals (field-filtered), and MNA parameters (hydrogen, soluble manganese, nitrate, chloride, sulfate, methane/ethane/ethene, and TOC). Groundwater field parameters (pH, temperature, dissolved

oxygen, ORP, salinity, conductivity, turbidity, alkalinity, ferrous iron, and hydrogen sulfide) were measured at each well and are presented in Table 4.6. Additionally monitoring wells MW-2S, MW-55S, MW-61I, MW-56D, MW-1, and MW-53S were analyzed for perchlorate.

Significant observations during sampling activities include:

- Prior to purging MW-55S, the water was a grey color and a petroleum odor was noticed. The PID reading was 1.6 ppm. There were no odors or elevated PID readings after purging and prior to collecting the groundwater sample from this well.
- A petroleum odor was observed and a PID reading of 73 ppm was recorded prior to purging at MW-55D. There were no odors or elevated PID readings after purging and prior to collecting the sample.
- A PID reading of 2.6 ppm was observed at MW-59S prior to purging. PID readings decreased to 0 ppm during purging and sampling.
- At the start of purging well MW-61I, the water was turbid (7,200 NTU) and brown in color. No odors or elevated PID readings were observed. After purging, the water was clear, and turbidity stabilized at 9 NTU.
- Prior to purging MW-101S, a petroleum odor was observed and a PID reading of 46 was recorded. The water was grey in color with a turbidity of 180 NTU. After purging, there were no noticeable odors or PID readings, and the water was clear with a turbidity reading of 5 NTU.
- At the start of purging well MW-102D, the water was turbid and brown in color. No odors or elevated PID readings were observed. After purging, the water was clear, and turbidity stabilized at 10 NTU.
- At the start of purging well MW-103D, the water was turbid (990 NTU) and light brown in color. No odors or elevated PID readings were observed. After purging, the water was clear, and turbidity stabilized at 9 NTU.
- At the start of purging well MW-104S, the water was turbid and grayish-white in color. No odors or elevated PID readings were observed. After purging, the water was clear, and turbidity stabilized at 7 NTU.

4.4 GEOLOGY

The borings completed during the Supplemental RI generally confirm the site geology described by the previous investigations (Metcalf & Eddy, Inc., 1996b; and Versar, Inc., 2000). The lithologic unit immediately underlying the site, the Columbia Group, was found to consist of predominantly fine to medium sand with lesser amounts of silt and clay. The Columbia Group was approximately 50 feet thick beneath the FFTA. During the Supplemental RI, an approximately 3-foot-thick silty clay layer was consistently found in the deep well borings (SB-102D, SB-103D, and SB-105D) at an elevation corresponding to sea level (approximately 25 feet deep). Well boring and construction logs are presented in Appendix E.

A review of the construction logs for the existing monitoring wells indicates four wells installed during the 1996 RI extended to a depth of at least sea level and were thus sufficiently deep to encounter this sea level clay layer [hereafter referred to as the Sea Level Clay, (SLC)]: MW-55D, MW-56D, MW-60I, and MW-61I. Of these wells, only the geologic log for MW-55D notes the occurrence of a clay layer at sea level. The geologic logs for MW-60I and MW-61I however, are incomplete and stop at depths of 16 feet and 17 feet, respectively, at least 10 feet above the expected depth of the SLC. Although the geologic log for MW-56D extends to a depth of 47 feet, it does not note the SLC. It is possible that the SLC was not noted in this boring because the split spoon sample interval was 5 feet, and the SLC is only approximately 3 feet thick. The wide areal distribution of the borings in which the SLC was found strongly suggests that it is continuous beneath the FFTA.

Consistent with previous findings, a silty clay layer was encountered in the three deep well borings at depths between 47 and 52 feet deep (an approximate elevation of -22 feet). This clay is interpreted as the upper aquitard of the Yorktown Formation based on the lithology and the agreement between the resultant observed thickness of the Columbia Group (approximately 55 feet) with its estimated regional thickness.

4.5 HYDROGEOLOGY

Groundwater levels were measured in the FFTA wells on March 3 and March 19, 2003 (see Table 4.5). The depth to groundwater at the FFTA was generally between 11 and 20 feet below ground (3.5 to 6.5 meters). The water levels on March 19 were approximately 0.5 (0.15 meters) foot higher than those measured on March 3. A qualitative screening of the data indicated a pronounced vertical hydraulic gradient. The deep monitoring wells consistently had a lower hydraulic head than the shallow wells. This is most clearly shown by the data from the MW-103 cluster of wells. On March 19, the water levels in the shallow, intermediate, and deep wells were 13.17 feet (4.01 meters), 16.13 feet (4.97 meters), and 19.01

feet (5.79 meters), respectively. These wells monitor the following depth intervals: 8.5 to 16 feet for the shallow well, 19 to 25 feet for the intermediate well, and 36 to 50 feet for the deep well. For the purpose of drawing groundwater elevation contour maps, the wells were grouped into shallow, intermediate, and deep classifications using the MW-103 wells as a guide. Wells with monitored intervals above approximately 15 feet were considered shallow, those with monitored intervals between approximately 15 and 30 feet were considered intermediate, and those with intervals that extended below 40 feet were considered deep. Table 4.5 shows the monitored intervals of the existing wells and their corresponding depth classifications. Most of the wells monitor the intermediate zone.

The groundwater elevation contours for the shallow, intermediate, and deep groundwater zones are illustrated in Figures 4-7 through 4-9, respectively. Due to the number of wells in each zone, there is a greater level of detail in the intermediate zone than in the shallow or deep zones. In the shallow and intermediate zones, the groundwater flow directions generally reflect the influence of surface topography. Most of the site area is flat but at the eastern side of the site, there is a steep slope into a ravine formed by the unnamed tributary to Little Mosquito Creek. In the shallow zone, groundwater in the immediate vicinity of the unnamed tributary flows east-southeast, following the slope of the ravine. In the intermediate zone, groundwater generally flows to the northeast. This direction coincides with the regional slope of the land surface towards Little Mosquito Creek, approximately 2,000 feet northeast of the site. In the immediate vicinity of the ravine, the flow becomes more easterly to southeasterly on the ravine's northern side and more northerly on its southern side, following the surface topography.

In the deep zone, the additional data provided by the three new deep wells have confirmed the northerly groundwater flow direction indicated in the 1996 RI (see Figure 3-3) (Metcalf & Eddy, Inc., 1996b). The available data do not indicate a significant eastward component to the groundwater flow in the deep zone. The groundwater elevation in MW-103D (10.29 ft/3.14 m), which is located near the top of the ravine slope, was almost identical with that in MW56D (10.34 ft/3.15 m), located approximately 275 feet to the west. The groundwater flow in the deep zone does not appear to be significantly influenced by the ravine of the unnamed tributary.

The horizontal hydraulic gradients beneath the FFTA are generally very low. In the shallow zone, limited data are available only for the part of the site with the steepest topographic gradient, near the unnamed tributary to Little Mosquito Creek. The horizontal hydraulic gradient from MW-103S, near the top of the ravine formed by the unnamed tributary, to WFF14GW-04, on the bank of the unnamed tributary, is approximately 0.03. In the intermediate zone, the horizontal hydraulic gradient across the central part of the site, from MW-54S to MW-57S, is approximately 0.004. Near the unnamed tributary, the horizontal gradient increases to approximately 0.02, as measured between MW-59S and MW-103I. These gradients are similar to those calculated from data generated in February 2000 (Versar, Inc., 2000). The horizontal

hydraulic gradient for the deep zone, as measured between MW-02D and MW-56D, is approximately 0.003.

Groundwater originating at or flowing through the source area (near MW-101S and MW-55S) flows generally northeast in the intermediate zone and north in the deep zone. Although there are no shallow zone wells in the immediate vicinity of the source area, groundwater in the shallow zone can be expected to flow in much the same general direction as the intermediate zone, but perhaps slightly more to the east towards the unnamed tributary.

To investigate the vertical component of groundwater flow beneath the FFTA, two hydrogeological cross sections were constructed along the orientations shown in Figure 4-10. Cross section A-A' extends from MW-102D, near the northern end of the site, to MW-14GW-05, on the bank of the unnamed tributary on the eastern side of the site. Cross section B-B' extends from MW-54S, just west of the source area, northeast to MW-104S, near the mouth of the unnamed tributary. These cross sections are displayed in Figures 4-11 and 4-12.

The cross sections illustrate the pronounced downward hydraulic gradient beneath the FFTA. This downward gradient is approximately 0.2 between MW-103S and MW-103D, which is at least an order of magnitude greater than the horizontal hydraulic gradient displayed at any of the three depth zones. The downward gradient indicates that the FFTA is located in an area of groundwater recharge.

As shown in Figure 4-12, there is little variation in the direction of the hydraulic gradient; it is steeply vertical at all locations. This is a reflection of the extremely low horizontal hydraulic gradient from MW-54S to MW-105D, as shown in this cross section. Figure 4-11 shows the increase in the horizontal hydraulic gradient near the unnamed tributary to Little Mosquito Creek. The vertical component of the gradient near the tributary remains downward.

The downward vertical flow of groundwater is interpreted to be impeded by the SLC and the upper aquitard of the Yorktown Formation. As discussed, previous aquifer pumping tests have confirmed the sealing efficiency of the Yorktown aquitard. The degree to which the SLC impedes the vertical migration of groundwater is not known.

The current data indicate that a significant portion of the groundwater flow in the shallow and intermediate zones is towards the unnamed tributary to Little Mosquito Creek. It is unlikely however, that the unnamed tributary is a significant groundwater discharge point. The vertical hydraulic gradient within the combined shallow/intermediate groundwater zone is downward, indicative of groundwater recharge. Based on the regional topography, Little Mosquito Creek and its adjacent wetlands are the likely groundwater discharge

points for the Columbia aquifer. It must also be noted that the unnamed tributary receives water from facility operations. During periods of low precipitation, this discharge water may comprise the total flow of the tributary. It is possible that the tributary serves as a source for groundwater recharge during such periods. Minor groundwater discharge may occur during periods of high precipitation when the water table rises sufficiently to intercept the tributary. This groundwater discharge however, would consist only of the shallowest groundwater (at and near the water table).

5.0 NATURE AND EXTENT OF CONTAMINATION

This section presents the results of the chemical analyses performed for surface soils, subsurface soils, and groundwater samples collected during this Supplemental RI. In addition, historical data for these media will also be included in this evaluation.

The data are summarized in tables for each sampling medium and in tables comparing analytical results to corresponding background results. The latter tables summarize the occurrence and distribution of results for each chemical (frequency of detection, minimum result, maximum result, and location of the maximum result). Individual sample results are presented in data summaries in a complete database format (Appendix A). The tabulated number of data points for each chemical varies depending on the number of results that were qualified as rejected or non-detected due to blank contamination as part of the data validation process.

5.1 USE OF HISTORICAL ANALYTICAL DATA

The work plan for the Supplemental RI (TtNUS, 2003a) and Sections 3.2 and 3.4 of this report detail the historical investigations and results at the FFTA. As presented in the previous sections, the 1996 RI included the collection and analysis of samples from other features located near the FFTA (WWTP sludge pile, debris piles, etc.) that are not considered in this Supplemental RI. The usability of the data and summary of results for each medium are discussed in this section.

5.1.1 Data Usability

Analytical data from historical investigations was verified as to validation status and the interpretation of qualifiers attached to individual results. Data qualified as rejected (R) or false positives attributable to blank contamination (B) were removed from further consideration. Data qualified as estimated (J) or biased high (K) or low (L) were retained. An assessment of data usability was performed to determine if detection limits were adequate to meet the project objectives defined in the work plan (TtNUS, 2003a). A review of the historical data for the FFTA indicates that these data are acceptable.

5.1.2 Surface Soils

Eight surface soil samples were collected from the FFTA during the 1996 RI (SS01, SS02, SS03, SS04, SS07, SS08, SS09, and SS10). The validated analytical results for these samples are included in Appendix A. Table 5.1 presents a summary of analytical results for these samples and surface soil samples collected during the Supplemental RI. The 1996 RI sampling locations are shown in Figure 4-3.

The results for these samples will be considered in evaluating the nature and extent of contamination at the FFTA and in assessing human health and ecological risks.

5.1.3 Subsurface Soils

Seventeen subsurface soil samples were collected during the 1996 RI from six borings (SB4 through SB9) advanced at the FFTA. The soil borings were placed in areas of suspected source contamination and based on projected groundwater contamination and flow patterns. Samples were collected from the 5-, 10-, and 15-foot depth intervals within each boring except SB4. Samples were collected only at the 5- and 10-foot depth interval in this boring. The validated analytical results for these samples are contained in Appendix A. A summary of analytical results for these samples and Supplemental RI samples is presented in Table 5.2. The boring locations are shown in Figure 4-4. The results for these samples will be considered in evaluating the nature and extent of contamination at the FFTA and in assessing human health risks. Because these samples were collected at depths greater than 2 feet below ground surface, they do not present a risk to ecological receptors and will not be considered in the ecological risk assessment.

5.1.4 Groundwater

Thirteen monitoring wells at the FFTA site and an off-site background well (MW-34S) were sampled during the 1996 RI. All samples were analyzed for VOCs, SVOCs, polychlorinated biphenyls (PCBs), pesticides, and metals (total and dissolved). Twelve of the FFTA wells were resampled in 2000 (MW-1, which was considered damaged was not sampled) to provide additional data for the updated risk assessment. The samples were analyzed for VOCs, SVOCs, metals, pesticides, and general chemistry parameters. The data were subsequently validated by an independent, third-party validation subcontractor in accordance with EPA Region 3 guidelines and procedures (Metcalf & Eddy, Inc., 1996b; Versar, Inc., 2000).

The 1996 RI included a human health risk assessment addressing potential future residential use of the groundwater present at the FFTA. The results of that risk assessment indicated that potential carcinogenic and noncarcinogenic risks exceeded levels considered acceptable by the VADEQ and EPA (carcinogenic range 10^{-4} to 10^{-6} ; noncarcinogenic hazard index of 1) (Metcalf & Eddy, Inc., 1996b). No formal risk assessment was performed using the results from the 2000 sampling event. Groundwater data from this sampling event were used to perform a qualitative risk evaluation or sensitivity analysis by comparing the results to the 1996 risk assessment (Versar, Inc., 2000). Both of these reports also evaluated the occurrence and distribution of the major contaminants identified at the FFTA, and based on that evaluation, estimated groundwater contaminant plumes for the most significant contaminants. These projected groundwater plumes are presented and discussed in Section 3.4.3.

Additional monitoring wells were installed during the Supplemental RI to augment the existing monitoring points. These new wells and the 12 existing wells were sampled in March 2003 and provide the most recent and most comprehensive data set available for the FFTA. Because the analytical data from the 1996 RI and the 2000 sampling event represent less complete data sets, do not represent current conditions, and because the samples were collected using varying sampling methods, the historical data will not be used for risk assessment purposes. The historical data will be used only for qualitative discussions of apparent contaminant patterns over time.

5.2 COMPILATION OF HISTORICAL AND SUPPLEMENTAL RI DATA

The Supplemental RI was performed to fill the data gaps identified in the work plan (TtNUS, 2003a). The Supplemental RI data, combined with the acceptable historical data discussed above, form the complete data set used to evaluate the nature and extent of contamination and determine risks to human health and the environment. The sample analytical results presented in Appendix A of this Supplemental RI contain the results from the historical data combined with the Supplemental RI data.

5.2.1 Supplemental RI Data Usability

All analytical data from the Supplemental RI were validated following EPA Region 3 and national functional guidelines. Data that are qualified as rejected (R) or false positives attributable to blank contamination (B) were removed from further consideration. Data that are qualified as estimated (J) or biased high (K) or low (L) were retained. Appendix F contains the validation reports and supporting documentation for the Supplemental RI data. An assessment of data usability was performed to determine if detection limits were adequate to meet the project objectives defined in the work plan (TtNUS, 2003a). Based on the data usability review, the analytical data for the Supplemental RI is acceptable except for the instances stated above (data qualified B or R).

5.2.2 Surface Soils

Eighteen surface soil samples were collected at 25- and 50-foot intervals along vectors emanating from the suspected source area (former fire training pit) during the Supplemental RI. Sample locations are shown on Figure 4-3. These locations were selected to provide additional data about the extent of contamination within and immediately adjacent to the suspected source area. Details of this sampling event are presented in Section 4.2 of this report. Samples were analyzed for VOCs (including MTBE), SVOCs, PCBs, metals, TOC, pH, and grain size. In addition, due to the potential for formation of chlorinated dioxins during site activities, surface soil samples were also analyzed for dioxins and furans.

The 18 samples collected during the Supplemental RI combined with the 10 surface soil samples from previous investigations detailed in Section 5.1.2 of this report form the data set used to evaluate the nature and extent of surface soil contamination at the FFTA. A summary of analytical results for the combined data set is presented in Table 5.1.

5.2.3 Subsurface Soils

Thirty-seven subsurface soil samples were collected from 18 borings during the Supplemental RI. The borings were located at the same 25- and 50-foot intervals along vectors emanating from the suspected source area described above for surface soil samples. Sample locations are shown on Figure 4-5. As detailed in Section 4.2 of this report, subsurface soil samples were collected at two distinct intervals at each of the 18 boring locations. A sample was collected from a depth of 2 to 4 feet at each location (probable depth of the former fire training pit) and a second sample was collected based on the observations of field personnel. A third sample interval was also collected at location FTA-SU-101. Samples were analyzed for VOCs (including MTBE), SVOCs, PCBs, and metals. Analyses for dioxins and furans were also performed for the subsurface samples from location FTA-SU-101.

The 37 samples collected during the Supplemental RI combined with the 17 subsurface soil samples from previous investigations detailed in Section 5.1.3 of this report form the data set used to evaluate the nature and extent of subsurface soil contamination at the FFTA. A summary of analytical results for the combined data set is presented in Table 5.2.

5.2.4 Groundwater

The Supplemental RI field investigation included the drilling and construction of seven new monitoring wells and sampling new and existing wells as detailed in Section 4.3. Monitoring wells within the FFTA study area were shown on Figure 4-6. The groundwater samples were obtained via low-flow sampling techniques following appropriate EPA Region 3 protocol and guidelines and were analyzed for VOCs plus MTBE, SVOCs, PCBs, TAL metals, TAL dissolved metals (field-filtered), and MNA parameters (hydrogen, soluble manganese, nitrate, chloride, sulfate, methane/ethane/ethene, and TOC). Groundwater field parameters (pH, temperature, dissolved oxygen, ORP, salinity, conductivity, turbidity, alkalinity, ferrous iron, and hydrogen sulfide) were measured at each well. Additionally monitoring wells MW-25, -55S, -61I, -56D, -1, and 53S were analyzed for perchlorate.

The 28 samples collected during the Supplemental RI form the data set used to evaluate the nature and extent of groundwater contamination at the FFTA. A summary of analytical results for the groundwater data set is presented in Table 5.3. Table 5.4 presents filtered (dissolved) metals data. As discussed in

Section 5.1.4, historical groundwater sample results were used to formulate the CSM and are used for qualitative evaluation purposes and to support the evaluation of contaminant trends only.

5.3 BACKGROUND DATA COMPARISONS

5.3.1 Statistical Approach

Statistical methods presented in the FFTA work plan (TtNUS, 2003a) were applied to compare the concentrations of substances detected in FFTA samples to those found in background samples. Statistical methods adhere to published guidance (EPA, 2002a, 2000, 1998, 1996a, 1994a, 1994b, 1994d, 1992a, 1992b, 1989a, and 1989b; US Navy, 2000, 1999, 1998) and other literature references (Gilbert, 1993 and 1987; Gehan, 1965; and Millard and Deverel, 1988).

Section 4.1 of this report presents a discussion of the WFF background study conducted in 2003. Soil and groundwater data considered usable from the background study are applied in this section to compare results from FFTA samples and to evaluate whether compounds detected on site are related to site operations or attributable to background or anthropogenic conditions.

Background comparisons for metals were handled somewhat differently than background comparisons for organics. The background study found significant differences in metals concentrations between Bojac and Molena soils and between surface and subsurface soil within the Bojac soil series. Because the Bojac soils predominate at the FFTA, site-related surface and subsurface soil data for metals were statistically compared to background surface and subsurface soil data sets from the Bojac classification. The background study revealed significant differences in concentrations of organic substances between surface and subsurface soil but not between the two soil series. Therefore, background organic data from the two soil types were consolidated to yield a larger surface soil data set that was statistically compared to FFTA surface soil organic data. The background subsurface soil organic data were also consolidated from both soil types for comparison to FFTA subsurface soil organic data.

The background study included approximately 10 samples of each background soil type, which was the stated goal in the FFTA work plan to achieve adequate statistical power to detect small differences between background and site concentrations. However, individual circumstances for specific analytes can also impact statistical power, such as distributional shape, frequencies of detection, and whether the range of detection limits are low compared to the range of positive results, so statistical background test results for certain chemicals were qualified as not applicable on a case-by-case basis.

There were not sufficient upgradient groundwater samples to allow quantitative statistical comparisons to FFTA site-related groundwater samples, so data were compared on a qualitative basis only. Appropriate background groundwater samples were limited to the wells upgradient of the FFTA Site and only samples collected in 2003 were utilized because different groundwater sampling methods were used in earlier sampling rounds.

5.3.2 Data Pretreatment

Several data pretreatment steps were necessary prior to conducting statistical tests. For chemicals with at least one positive detection in each data set, a value of one-half the sample quantitation limit was assumed for non-detect (U qualified) results. Duplicate samples were averaged together and considered as one result. For duplicates, where one result was positive and the other result was a non-detect, the problem of calculating an average result arose whenever half the detection limit exceeded the positive result. In these situations, the positive result was used to represent the non-detect.

The WFF background study conducted in 2003 included screening for extreme values and anomalies. Findings were documented in the background study report. No suspect data were identified among the samples collected in the 2003 supplemental investigation, and data qualified unusable (R) or affected by blank contamination (B) were eliminated from consideration in this Supplemental RI.

Prior to conducting any background data comparisons, the underlying statistical distribution of data was determined for each detected chemical. The Shapiro-Wilk W-test or the Shapiro-Francia Test (EPA, 1992d) were performed to determine if the data set of chemical concentrations matched the shape of a normal or lognormal distribution. [The latter test is required if there are greater than 50 samples (EPA, 1992d, 1996a)]. Normally distributed data exhibit a characteristic "bell-shape" curve that is symmetrical, whereas lognormal data have a skewed shape with a longer tail at the high-concentration end. For each analyte, the W-test was performed once using the original data and once after data was converted to their logarithms. A 5 percent level of significance was used to determine if the data deviated from either hypothesized distribution. Associated W-test scores greater than the critical value indicate that the hypothesis of a normal or lognormal distribution cannot be rejected at a 0.05 level of significance, respectively.

The distributional shapes that were assumed in parametric background tests are listed in the background test results in Tables 5.5 and 5.6. W-test scores for distributional analysis tests for all substances detected in groundwater, surface soil, and subsurface soil are presented in Appendix G.

The shape of the data distribution determined which background comparison tests were applicable as summarized by the following decision scheme:

Case 1: If the W-test accepted the hypothesis of a normal distribution for both site and background, the parametric t-test as well as the set of nonparametric background comparison tests were conducted to decide whether site exceeded background.

Case 2: If the W-test rejected the hypothesis of a normal distribution for both site and background, but accepted the hypothesis of a lognormal distribution for both site and background, then only the set of nonparametric background comparison tests were conducted to determine that site exceeded background.

Case 3: If the W-test rejected both hypothesis of either a normal distribution or lognormal distribution for both site and background, then only nonparametric background comparison tests were conducted.

Special Cases for the upper tolerance limit (UTL) test: The UTL test was not used to decide globally that site exceeds background, but rather IF site data were first determined to exceed background based on the other statistical tests, then the information presented in the UTL test could be used to evaluate whether individual sample results might represent hot spots. If the W-test accepted the hypothesis of a normal distributional shape and the score reflected a better fit than for a lognormal shape, then the UTL test was based on the raw (not log-transformed) data. If the W-test accepted the hypothesis of a lognormal distributional shape and the score reflected a better fit than for a normal distribution shape, then the UTL test was based on the log-transformed data. If the W-test indicated that the data did not match either a normal or lognormal shape and there were at least 19 data points in the background data set, then the UTL was based upon substitution of the non-parametric 95th quantile of the background data set.

5.3.3 Background Comparison Tests

To determine if results of samples from the FFTA site are elevated relative to background sample results, an array of statistical procedures was applied that consist of three quantitative tests that look for overall differences between the entire populations of site and background data values; four quantitative tests that essentially look for hot spots; and two qualitative tests that examine only the frequency of detection (proportion of detected versus non-detected values in site versus background) but not the magnitude of values.

Each statistical test was run using a decision-making probability level (P-level) of 0.05, which means that, in situations where the test conclusion states that site-related results are greater than background, the chance of the test yielding a false conclusion caused by random variations in the data set is five percent or less. The overall conclusion (whether site results are greater than background) was assumed to be “yes” if any one of the quantitative tests concluded that site data are elevated above background. If no conclusion could be reached for any of the quantitative tests (e.g., if the assumptions necessary to run each of the various tests were not valid), then the overall decision was based on the conclusions of the qualitative tests alone. Further information regarding each statistical test is presented below:

- The means of the two data sets were compared if both site and background matched a normal distribution. If the site and background data exhibit equal standard deviations (based upon Bartlett's test for equal variances), then the student's t-test was applied; otherwise, Satterthwaite's t-test was performed to see if the site mean is greater than the background mean. The t-test is valid only if at least 85 percent of site data and 85 percent of background data are positive detects, there are at least three sampling points in each data set, and the pooled standard deviation is not zero.
- Nonparametric statistical tests, which do not require underlying assumptions regarding equal data distributions, were also applied in each case. The Mann-Whitney U-test was used to determine whether the site and background data are from populations with identical medians and rank distributions. The Mann-Whitney test involves combining the two data sets, ranking results from smallest to largest, and evaluating whether the two sites have a similar distribution of data within the range of low to high ranks. If more than 40 percent nondetected results were present in either the site or the background data set or when multiple levels of detection limits are present, a different statistically valid test, Gehan's test, was substituted because recent guidance (EPA, 1992c) indicates that the Mann-Whitney test is not valid in the aforementioned situations. (Gehan's test is statistically equivalent to the Mann-Whitney Test if all results are positive.) For either of these tests to work, not all data points can be tied and there must be at least two background data points. The Mann-Whitney U test and the Gehan test statistics were computed using appropriate score adjustments for tied values and a normal approximation when sufficient data points were available; whereas, an exact computation of probabilities was used in the situations where there were very few (for example, less than eight) data points.
- A 95 percent upper tolerance limit (UTL) test was performed strictly to provide supplementary information to assist in locating potential hot spots of a magnitude exceeding 95 percent of the background population. Because the UTL test is designed to compare only one site-related sample to background, erroneous conclusions can occur if this test is relied upon when the site data consist of a larger number of samples, since in such cases the probability of one or more of the site-related

samples exceeding the UTL is much greater than 5 percent even when the site and background data belong to the same underlying statistical population. Therefore, in accordance with Navy guidance, the UTL test was only considered as a source for qualitative information and other statistical tests were relied upon as the basis for decisions regarding whether the site population exceeds background.

- The 95 percent UTL is defined as the calculated upper limit that, on the average, is expected to include 95 percent of the background population (EPA, 1992a). If the background data were determined to match the shape of a normal or lognormal population, then the limit was calculated using the t-distribution or K-value and the appropriate normal or log-transformed mean and standard deviation from the background data set. For this test to be valid, the background data set is required to be comprised of at least 85 percent detects and at least three data points.
- A substitute procedure for the 95 percent UTL test, called the 95 percent quantile test, was employed to provide supplementary information to assist in locating potential hot spots if the background data were not in the shape of a lognormal or a normal population. For the quantile test to be valid, at least 19 background data points are required, no detection limit can be greater than the UTL, and at least 10 percent of the data points must be detects in the background data set.
- The upper ranks test (EPA, 1992c, 1996d) is another hot spot test. This test combines the site and background data into one set and determines whether the major portion of a subset of the largest detected results is comprised chiefly of site data rather than an equal mixture of site and background. In this procedure, the probability is calculated that k or more samples from the largest r data points in the combined data set are comprised of site data, assuming that the site and background populations are equal. In the event that there is less than a five percent chance that this could happen if the populations are indeed the same, then the test concludes that there is a hot spot comprised of samples from the area of interest. This test was used for making decisions regarding background.
- In the event that none of the above quantitative statistical tests yielded a definite “yes” or “no” decision, a test of proportions was used to determine if the percentage of positively detected results is greater in the site data versus the background data. When only a very small portion of results are detected (less than 10 percent), this test is recommended (EPA, 1996d, 1989b). The test is routinely applied using a normal distribution approximation to the probability that site is above background but is not considered valid when fewer than five samples are detected in either site or background. To reach a confident decision in such cases, a generalized version of the test of proportions, called the Fisher Exact Test, is required (Brownlee, 1965). This test was applied to all situations because it calculates the exact probability for all combinations of possible outcomes and gives a probability level

for the condition where the observed frequency of site detects is greater than background, given the number of samples involved.

The following provides a breakdown of the decision scheme applied to determine which statistical tests were used:

Case 1: If the W-test accepted the hypothesis of a normal distribution for both site and background, then the W-test using the logarithmic conversion was not applicable and parametric as well as nonparametric background comparisons were conducted.

- For this case, the parametric t-test was performed to compare the arithmetic means of site and background only if there were greater than 85 percent detects in the site data and greater than 85 percent detects in the background data. A Satterthwaite adjustment of the t-test was used if the standard deviations between site and background data were found to be significantly different from one another (criteria based on Bartlett's Test for equal variances).
- For Case 1 (but also applicable to Cases 2 and 3, as discussed later), a nonparametric test was also selected to compare the ranks of the overall populations, based on either the Mann Whitney test or Gehan's test. The prevalence and magnitude of non-detects can bias the outcome of the Mann Whitney test, but not so for Gehan's test. Therefore, Gehan's test replaced the Mann Whitney test if any of the following circumstances occurred:
 - a. if more than 40 percent non-detects were present in either the site data set or the background data set;
 - b. if the number of non-detect data points from the site data that are greater than both data set's lower 25th percentile for positive detections is greater than 10 percent of the total number of detects in the site data;
 - c. if the number of non-detect data points from the background data that are greater than both data set's lower 25th percentile for positive detections is greater than 10 percent of the total number of detects in the background data;
 - d. if the ratio of the highest detection limit to the lowest detection limit among all data groups combined was greater than 3.5;

e. if greater than 15 % of the data points in the site data set consisted of non-detects that were less than the 25th percentile of the subset of non-detected site data but greater than the 25th percentile of the subset of non-detected background data (a relative shift in the distribution of the two data sets' non-detects occurring in a low percentile range). The same test was also performed for the converse situation (more than 15 percent of background data consist of nondetects whose magnitude was less than the 25th percentile of non-detected background data but greater than the 25th percentile of non-detected site results). The same tests were also performed at the 75th percentile and 95th percentile of site and background data sets, such that evaluating all of these tests together served to identify any significant shifts in the relative distribution of non-detects compared between the site population and background population.

- Technically, although Gehan's test could be substituted for the Mann Whitney test in all circumstances, the Mann Whitney test is more familiar to scientists and available in software, so the use of the Mann Whitney test has been retained because it is easier for a third party to independently verify the calculations and test results.
- For Case 1 (but also applicable to Cases 2 and 3, as discussed later), a second nonparametric test was selected to compare the upper ranks (i.e., the upper percentiles) of the site data versus background data. This test has been labeled as "upper ranks" to avoid confusion with a separate test for upper tolerance limits (UTL) that also involves quantiles for the nonparametric case. The term "quantile test" is also used for the upper ranks test. This test was computed multiple times, starting with the highest concentration (rank) data points, then repeating the test after including the next lower concentration sample results, thereby successively expanding the size of the upper rank subsets to include lower percentiles of the combined site and background data. The upper ranks tests were concluded when the highest non-detected result was encountered or when the maximum size subset of site data was found that had at least a probability (P-level) of 0.05 that the upper ranks of this subset of data is significantly greater than background.
- For Case 1 (but also applicable to Cases 2 and 3, as discussed later), if each of the applicable test results (t-test, Mann Whitney, Gehan, and Upper Ranks) were considered not applicable ("NA") due to insufficient sample size or other limitations, then a test of proportions was considered applicable to determine if the percentage of positively detected results was statistically greater in the site data versus the background data. The test of proportions was performed one of two ways: The first method is based upon combinatorial probabilities calculated using Fisher's exact test if there were less than 5 detects in the site data set, less than 5 detects in the background data set, less than 5 non-detects in the site data set, or less than 5 non-detects

in the background data set. The second method applied if there were 5 or more detects and non-detects for each data set, in which case a normal approximation to the probability of proportions was computed using the z-test. These results were used to decide site is greater than background only if other tests were "NA".

- For Case 1, if a decision was first reached that site concentrations were greater than background based on the tests presented earlier, then the 95 percent upper tolerance limit (95 % UTL) was considered applicable. The UTL test was not used to decide globally that site exceeds background, but rather IF site data exceeded background, then the information presented in the UTL test could be used to evaluate whether individual sample results might represent hot spots. Following previous risk assessment recommendations from EPA Region 3, the UTL was computed in two different ways, using formula presented in EPA, 1992a: In the case where there were less than 8 samples in the background data set, the UTL was calculated assuming a 50 percent coverage of the 95 percent UTL, so that formula included the student's t-value times the standard deviation. In the case where there were 8 or more samples in the background data set, the UTL was then calculated in a more conservative manner assuming 95 percent coverage of the 95 percent UTL, so that the formula included the K-value times the standard deviation. Note that the UTL test has an unacceptable rate of false positives if there are many site samples, and the existence of one or more exceedances in a data set comprised of more than three site samples does not by itself necessarily indicate that site exceeds background.

Case 2: If the W-test rejected the hypothesis of a normal distribution for both site and background, but accepted the hypothesis of a lognormal distribution for both site and background, then only the set of nonparametric background comparison tests were conducted to determine that site exceeded background. The following tests were performed for Case 2:

- Either the Mann Whitney or Gehan's test was performed, as discussed above.
- The upper ranks test was performed, as discussed above.
- The t-test was not performed.
- The test of proportions was performed and considered in making decisions only if the Mann Whitney, Gehan, and upper ranks tests were all not applicable ("NA").
- If a decision was first reached that site concentrations were greater than background based on the tests presented earlier, then the 95 percent upper tolerance limit (95 % UTL) was considered

applicable. The UTL test was not used to decide globally that site exceeds background, but rather IF site data exceeded background, then the information presented in the UTL test could be used to evaluate whether individual sample results might represent hot spots.

Case 3: If the W-test rejected both hypothesis of either a normal distribution or lognormal distribution for both site and background, then only nonparametric background comparison tests were conducted. The following tests were performed for Case 3:

- Either the Mann Whitney or Gehan's test was performed, as discussed above.
- The upper ranks test was performed, as discussed above.
- The t-test was not performed.
- The test of proportions was performed and considered in making decisions only if the Mann Whitney, Gehan, and upper ranks tests were all not applicable ("NA").
- If a decision was first reached that site concentrations were greater than background based on the tests presented earlier, then the 95 percent upper tolerance limit (95 % UTL) was considered applicable. The UTL test was not used to decide globally that site exceeds background, but rather IF site data exceeded background, then the information presented in the UTL test could be used to evaluate whether individual sample results might represent hot spots. In this case, the UTL test was based upon the non-parametric 95th quantile and was only possible if there were at least 19 data points in the background data set.

For this project, computer algorithms were employed to automatically perform applicable statistical tests wherever the necessary underlying assumptions are valid. This approach allows the use of any valid test result that identifies that concentrations from one data set are elevated, regardless of the outcome of unrelated statistical tests.

5.3.4 Surface Soil Background Comparison Results

Table 5.5 compares the FFTA site-related surface soil data to background surface soil data. Surface soil data were demonstrated to be above background for lead, fluoranthene, and methylene chloride. With lead, the upper ranks test revealed that seven of the highest ranked concentrations in the site data set were greater than any values in the background data set, which has a probability of occurring only 4 percent of the time in statistically equivalent populations. Therefore, these findings are significant and

point to the need for evaluating potential hot spots in these 7 samples. However, elevated lead results were not observed in the majority of samples because the Mann Whitney test did not reveal an overall difference between the two populations (as would occur if means or medians were different).

In the case of fluoranthene, Gehan's test indicated that there was an overall difference in the relative ranks of the site and background populations, but because data did not follow a normal or lognormal distribution the means of the two populations could not be directly compared. In addition, the frequency of detection was borderline significant, with a 0.06 probability level (P-level) associated with 7 out of 25 site detections compared to only 2 out of 26 background detections. Examination of data showed that the mean was influenced most heavily by one high concentration sample, which was an order of magnitude greater than other results.

For methylene chloride, data should be interpreted with caution because this chemical is a common laboratory solvent and detected FFTA surface soil concentrations of methylene chloride were generally less than the background sample quantitation limits.

Surface soil background test results were inconclusive for antimony, mercury, silver, and thallium. This was due to several of the results being nondetected or rejected, which reduced the effective size of the data sets that were compared. However, for each of these metals, mean FFTA surface soil concentrations were all less than 0.5 mg/kg. Similarly, the low frequency and low level of detections interfered with statistical tests and no conclusions could be reached for other organic compounds.

5.3.5 Subsurface Soils Background Comparison Results

Table 5.6 compares the FFTA site-related subsurface soil data to background subsurface soil data. Subsurface soil data were demonstrated to be above background only for bis(2-ethylhexyl) phthalate. Gehan's test indicated that there was an overall difference in the relative ranks of the site and background populations, but because data did not follow a normal or lognormal distribution the means of the two populations could not be directly compared. From a qualitative standpoint, the frequency of detection was also greater, with nearly half of FFTA subsurface soil samples representing positive detections compared to less than 15 percent of background samples. However, bis(2-ethylhexyl) phthalate data should be interpreted with caution because this chemical is a common laboratory solvent and detected FFTA subsurface soil concentrations in 16 out of 21 samples were generally less than the background sample quantitation limits.

Subsurface soil background test results were inconclusive for antimony, beryllium, cadmium, and thallium. This was due to several of the results being nondetected or rejected, which reduced the effective size of

the data sets that were compared. However, for each of these metals, mean FFTA subsurface soil concentrations were all less than 0.5 mg/kg. Similarly, the low frequency and low level of detections interfered with statistical tests and no conclusions could be reached for other organic compounds.

5.4 NATURE AND OCCURRENCE OF CONTAMINATION

5.4.1 Surface Soils

Results of the background tests, as shown in Table 5.5, indicate that a limited number of contaminants were detected in surface soil samples from the FFTA at levels exceeding background concentrations. For metals, only lead was identified as exceeding background levels; statistical tests were inconclusive for antimony, mercury, silver, and thallium. Table 5-7 presents the occurrence and distribution of contaminants detected in the background and site-specific surface soil samples.

Lead, as shown in Table 5.7, was detected in background samples at concentrations ranging from 0.571 mg/kg to 16.9 mg/kg and in site samples from 3 mg/kg to 40.8 mg/kg. The highest site lead concentration, 40.8 mg/kg, was detected in sample FFTA-SS-117 (see Figure 5-1). Six other site surface soil samples had lead concentrations greater than the maximum background level. The lead concentrations ranged from 17.1 mg/kg to 33.8 mg/kg in these samples. The lead results and location for these samples are shown on Figure 5-1. As can be seen in that figure, the elevated data points appear to form a cluster near the suspected location of the former fire pit.

Although the statistical comparison of the background and site data sets was inconclusive for antimony, mercury, silver, and thallium, the site data do not suggest that these metals are major site contaminants. Antimony was detected in three site samples at levels greater than the maximum background concentration, 0.44 mg/kg. The highest site-related detection was 1.5 mg/kg (sample FFTA-SS-106) and the other two detections (0.45 mg/kg at FFTA-SS-104 and 0.53 mg/kg at FFTA-SS-105) were close to the maximum background concentration. Mercury was detected in eight site samples, but the maximum concentration, 0.079 mg/kg, was less than the maximum background concentration, 0.21 mg/kg. Silver and thallium were each detected in only one site sample. Silver was detected at 0.19 mg/kg in sample FFTA-SS-115 and in three background samples at concentrations ranging from 0.04 mg/kg to 0.13 mg/kg. Thallium was not detected in any of the background surface soil samples and was detected at 0.18 mg/kg in only one site sample, FFTA-SS-115.

For PAHs, background tests indicate only flouranthene in FFTA surface soils above background levels; results for several other PAHs were inconclusive. Fluoranthene was detected in seven site surface soil samples but only in sample FFTA-SS-101 at levels (6,000 ug/kg) greater than the maximum background

concentration. This sample also contained the highest levels of other PAHs detected at the FFTA. Although PAHs were detected at relatively low frequencies and concentrations in other site samples, a review of total PAHs concentrations indicates a possible contaminant pattern centered around sample FFTA-SS-101, which contained 32,750 ug/kg of total PAHs. Total PAH values are presented in Table 5.1 and are illustrated on Figure 5-1.

Analyses for PCBs showed only one positive result. Aroclor-1260 was detected at an estimated 630 (J) ug/kg at FTA-SS-117. No other PCBs were detected during the 1996 RI or the Supplemental RI.

For VOCs, which are not typically considered anthropogenic contaminants, methylene chloride was the only compound for which statistical tests indicated that site contamination above background levels; methylene chloride was detected in three samples at concentrations ranging from 5 J ug/kg to 9 J ug/kg. Methylene chloride is a common laboratory contaminant that was rejected due to blank contamination in most of the 1996 RI and the Supplemental RI soil samples. The three results that were not rejected do not indicate that methylene chloride is a major site contaminant. Similarly, acetone is also a common laboratory contaminant that was detected at estimated levels in three of the site samples. 2-Hexanone was detected in only one soil sample at an estimated concentration of 6 ug/kg.

Other VOCs detected on site include 1,2,4-trimethylbenzene, 1,3,5-trimethylbenzene, methyl acetate, tetrachloroethene, and toluene. All of these compounds were infrequently detected at estimated concentrations near or less than the detection limits. Reported concentrations for these compounds are shown on Figure 5-1. Although the levels and frequencies of detection are low, the presence of these compounds, in general, is consistent with previous soil gas findings that VOC contamination existed within the central portion of the site associated with the former fire training pit

The FFTA surface soil samples were analyzed for dioxins and furans because of the history of burning petroleum-related products at the site. The assessment of chlorinated dioxin and furan congeners was performed using the latest toxicity equivalency factors (TEFs) that relate the toxicity of various congeners to the toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) (WHO, 1998). Using two sets of TEF values, one for human and mammalian receptors and the other for avian receptors, concentrations of individual dioxin and furan congeners were converted to an overall toxicity equivalent concentration (TEQ) for each sample, which represents a concentration of 2,3,7,8-TCDD associated with the same overall risk. Only detected congeners were summed within each sample to yield a total TEQ of 2,3,7,8-TCDD concentration for that sample. Values qualified during data validation as estimated maximum possible concentrations (EMPCs) did not pass the required EPA Region 3 criteria for compound identification and were omitted from TEQ calculations. Appendix A contains the full data set for the dioxin and furan analysis, while Table 5.1 and other risk assessment tables contain only the overall TEQ values for dioxins/furans.

Background surface soil samples were also analyzed for dioxins and furans, and a comparison of the background total 2,3,7,8-TCDD TEQs to site data (see Table 5.5) indicates that site contaminant levels are not elevated above background levels. Table 5.7 presents the range of background and site-related total 2,3,7,8-TCDD TEQs for human receptors. The range of detections for the background and site samples are similar and the site mean (0.00421 ug/kg) is less than the background mean (0.0048 ug/kg).

5.4.2 Subsurface Soils

Results of the background tests for subsurface soils, as shown in Table 5.6, indicate that no metals in subsurface soil samples from the FFTA were identified as exceeding background levels. Tests were inconclusive for antimony, beryllium, cadmium, and thallium. Table 5.8 presents the occurrence and distribution of contaminants detected in the background and site-specific subsurface soil samples. Antimony was detected in eight site subsurface soil samples. The maximum detected concentration, 3.4 mg/kg, was detected in sample FTA-SU-101-1617. This sample was collected from a depth of 16 to 17 feet from a 0.5 feet thick layer of black material described as being “charred or charcoal-like” with a strong fuel odor. Antimony levels detected in the other seven samples ranged from 0.17 mg/kg to 5.7 mg/kg. Figure 5-2 shows the location of FTA-SU-101-1617.

Beryllium was detected, in general, throughout the site. Thirty-seven samples contained detectable levels of beryllium. The highest concentration detected in site subsurface soils was 0.44 mg/kg, which is similar to the maximum background concentration of 0.3 mg/kg. Only four samples (FTA-SU-103-0204, FTA-SU-110-0204, FTA-SU-113-0709, and FTA-SU-116-0709) contained beryllium concentrations (0.32 mg/kg to 0.44 mg/kg) above the maximum background concentration. The occurrence and distribution of beryllium in the subsurface does not suggest that this metal is a major contaminant at the FFTA. Similarly, thallium was detected in 16 site samples at concentrations ranging from 0.17 mg/kg to 0.4 mg/kg. Most positive detections were qualified as being biased high due to exceedance of quality control criteria. Cadmium was only detected in three of the site subsurface samples with concentrations ranging from 0.044 mg/kg to 0.46 mg/kg. The maximum detected concentration, 0.46 mg/kg, was the only detected concentration greater than the maximum background concentration, 0.22 mg/kg.

Statistical comparisons of site to background lead levels indicate that overall site subsurface lead levels are within expected background levels. However, lead was detected in three subsurface soil samples above maximum background concentrations. Samples FTA-SU-101-0204, FTA-SU-101-1617, and FTA-SU-104-0204 contained lead at 91.7 mg/kg, 80.4 mg/kg, and 26.2 mg/kg, respectively. Surface soil samples collected from these two locations also contained the highest surface soil lead levels.

As can be seen in Table 5.6, background subsurface comparison tests were inconclusive or considered statistically invalid for organic compounds. Table 5.8 presents the occurrence and distribution of organic contaminants detected in FFTA subsurface soil samples. Although most SVOC and PAH compounds were infrequently detected, sample FTA-SU-101-1617 contained the highest levels and number of contaminants. PAH levels in this and other subsurface soil samples are shown on Figure 5-2. The samples collected from the 16 to 17 foot and 7 to 8 foot depth intervals in boring FTA-SU-101 contained the highest levels and number of contaminants. Sample FTA- SU-105-0911 (collected from the 9 to 11 foot horizon) contained the next highest levels and numbers of contaminants. This pattern of contamination is consistent with the PAH contamination noted in surface soils, although levels are higher and less dispersed. No PCBs were detected in subsurface soil samples from the FFTA site.

VOCs that defined the nature and extent of groundwater contamination in previous studies or that were detected at a high level of frequency in site subsurface soils include 1,1,1-trichloroethane (TCA); 1,1-dichloroethane (DCA); 1,1-dichloroethene (DCE); cis,-1,2-DCE; 1,2,4-trimethylbenzene; 1,3,5-trimethylbenzene; benzene; ethylbenzene; tetrachloroethene (PCE); toluene; and xylene. The locations and concentrations of these compounds are shown on Figure 5-2. The samples with VOC contamination were all collected from borings advanced near the center of the former fire training pit. The highest number and levels of VOC contaminants were detected in samples FTA-SU-105-0911 (collected at the 9 to 11 foot horizon) and FTA-SU-101-0708 (collected from the 7 to 8 foot depth horizon).

Three subsurface soil samples from the FFTA were analyzed for dioxins and furans. The subsurface soil samples collected from boring FTA-SB101 were analyzed for dioxins and furans because, as noted previously, a black “charred or charcoal-like” material was encountered at the 16 foot depth in this boring. The full analytical results for these samples are contained in Appendix A. To assess the significance of these compounds at the site, total 2,3,7,8-TCDD TEQs were calculated for each sample and are presented in Table 5.2. In accordance with the work plan and as described in Section 7 of this report, two separate TEQs, one for avian ecological receptors and one for mammal or human receptors, were calculated. Background subsurface soil samples were not analyzed for dioxins and furans. The range of total 2,3,7,8-TCDD TEQs for human receptors is shown on Table 5.8. The highest TEQ, 0.01059 ug/kg, was detected in the sample obtained from the 16 to 17 foot depth horizon. The 2,3,7,8-TCDD TEQs for the 2 to 4 foot and 7 to 8 foot depth samples were 0.00775 ug/kg and 0.00208 ug/kg, respectively.

5.4.3 Groundwater

Table 5.3 presents a summary of analytical results for groundwater samples at the FFTA and in site-specific background (upgradient) samples collected during the Supplemental RI. The occurrence and distribution of the contaminants is presented in Table 5.9. There are insufficient background samples to

conduct a statistical background comparison for groundwater. Background results will be used to qualitatively evaluate site-specific data.

Arsenic and lead were identified as site-related groundwater contaminants in the 1996 RI, primarily based on detections at monitoring wells MW-54S and MW-55S. Arsenic was detected in three of the FFTA monitoring well samples (FTAMW-55S, FTAMW-55D, and FTAMW-61I) and one of the background samples (MW1-20030303) during the Supplemental RI. Analytical results are shown on Figure 5-3. Filtered groundwater samples were also collected from all wells during the Supplemental RI to evaluate the presence of dissolved metals. Analytical results for the filtered samples are presented in Table 5.4. Dissolved arsenic was detected in the same three site monitoring wells, with the highest concentration (25.6 ug/L) detected at MW-55S. Similar to arsenic, monitoring well FTAMW-55S contained the highest lead levels in unfiltered and filtered samples, 73.2 ug/L and 59.6 ug/L, respectively. Total lead was identified only in samples collected from monitoring wells FTAMW-55S, FTAMW-55D, and FTAMW-58S. Dissolved lead was detected only in samples collected from MW-55S and MW-61I. Samples from these wells also contained the highest levels of iron and manganese found at the site.

Several SVOCs were detected during the Supplemental RI; however, phenanthrene, selected as a groundwater contaminant of potential concern (COPC) in 1996 and confirmed in 2000, was not detected. Naphthalene, detected at levels up to 2,000 ug/L in 1996 and 270 ug/L in 2000 in MW-55S, was present in four wells at concentrations ranging from 21 ug/L to 66 ug/L (MW-55S, MW-56D, MW-58S, and MW-61I). These four wells are located within and immediately downgradient of the suspected source area (see Figure 5-3). The highest levels were recorded in MW-61I. Other SVOCs detected during the Supplemental RI included phenolic compounds in MW-55S and MW-55D (see Table 5.3).

Historical results for VOCs indicate contamination, in general, extending from the fire training pit area (MW-55S) in a northerly and easterly direction. Although other VOCs were detected during the previous investigations, the groundwater contamination was characterized by projected benzene, toluene, and cis-1,2 DCE plumes. Results from the Supplemental RI show similar contaminants as previous investigations (see Table 5.3). However, the compounds selected as COPCs in 1996 were detected at lower levels during the recent sampling. Figure 5-3 shows the concentrations of selected VOCs detected in monitoring well samples collected and analyzed during the Supplemental RI. The VOCs shown were selected based on their historical use to define groundwater contamination, potential human health risks, and frequency of detection.

The 1996 RI and the 2000 sampling event projected a benzene plume centered around MW-61I, see Figures 3-5 and 3-8. Figure 5-4 shows the benzene isoconcentrations using the 2003 Supplemental RI data. As can be seen in all three figures, the highest concentrations of benzene have remained in the area

around MW-61I. However, benzene levels decreased from 100 ug/L in 1996 to 31 ug/L in 2000 and to 28 ug/L in 2003. The northern extent of benzene in groundwater as indicated in the investigations is in the area of MW-57S, which current data indicate contains benzene at 3 ug/L. The south-southwestern extent, as defined in the current data is at MW-55S (2 ug/L). A low level of benzene was detected at MW-2S (2 ug/L) in 1996, but benzene has not been detected in this well since that event.

Toluene was detected in MW-55S and MW-55D during the 1996 and 2000 sampling events. The highest levels were detected in MW-55S (1,300 ug/L to 1,400 ug/L) with significantly lower levels in MW-55D (e.g., 32 ug/L in 2000). Figures 3-6 and 3-9 show the interpreted isoconcentrations of toluene for these two sampling events. Results for the Supplemental RI show toluene still present in MW-55S, but not MW-55D. Concentrations in MW-55S were substantially less than previously reported (83 ug/L). The only other Supplemental RI groundwater sample with a positive detection of toluene was collected from MW-101S (12 ug/L), which is located approximately 50 feet south of MW-55S. There is no indication that toluene has migrated further northeast towards MW-61I, which generally shows the highest levels of VOCs detected on site. Although highly interpretive, a projected toluene plume based on these two detections is shown on Figure 5-5.

The most extensive groundwater plume projected during the 1996 RI and 2000 sampling events was for cis-1,2-DCE. The 1996 RI reported cis-1,2-DCE concentrations ranging from 1 ug/L to 3,000 ug/L and projected a cis-1,2-DCE plume centered around MW-61I and extending from the fire training pit (MW-2 and MW-55) to the northeastern portion of the site (MW-57S). The cis-1,2-DCE plume projected based on the 1996 RI data is shown on Figure 3-7. The 2000 resampling showed a similar pattern of contamination. However, as shown on Figure 3-10, concentrations were reported at lower levels and the plume was projected to have decreased in extent in the southern portions (upgradient) of the site and shifted to the east in the northeastern portions of the site. Results from the Supplemental RI sampling indicate a similar trend and further decrease in contaminant levels. Cis-1,2-DCE was detected in 10 of 21 samples at concentrations ranging from 1 ug/L to 460 ug/L. MW-61I again had the highest level of this compound (460 ug/L); the levels detected in 1996 and 2000 were 3,000 ug/L and 1,000 ug/L, respectively. A projected cis-1,2-DCE plume, based on the Supplemental RI data is presented in Figure 5-6. The highest concentrations continue to extend from MW-61I northeastward to MW-56D (360 ug/L) and MW-57S (110 ug/L), but was detected at only 1 ug/L in MW-105D screened in the deeper portion of the aquifer at this location. In addition, cis-1,2-DCE was not detected in the deep far downgradient well MW-104S but was detected in shallow wells to the east (MW-103I and 14GW-05). In 2000, the eastern wells MW-14GW-04 and MW-14GW-05 had reported concentrations of 47 ug/L and 5 ug/L, respectively. Cis-1,2-DCE was not detected in a sample collected from MW-14GW-04 during the Supplemental RI and was reported at 19 ug/L in a sample from MW-14GW-05.

Previous analytical results for 1,1,1-TCA indicated another possible plume, although not mapped, extending from MW-55S to MW-61I. 1,1,1-TCA was detected in 6 of 12 samples in the 2000 resampling with the highest levels at MW55S (510 ug/L). Results from the Supplemental RI indicate an east-west oriented 1,1,1-TCA plume as shown in Figure 5-7, with the highest concentrations located at MW-58S (340 ug/L) and MW-61I (240 ug/L). Levels at MW-55S have decreased to 46 ug/L. The plume extends northeastward from MW-58S and MW-61I toward MW-56D (210 ug/L) and MW-57S (84 ug/L) as well as eastward to MW-103I (19 ug/L). The shallow and deep wells at MW-103 do not show 1,1,1-TCA. The data indicate that the plume may be decreasing in concentration and size while migrating to the north and northeast. Other VOCs detected also show this northeasterly trend from the area near MW-55S to MW-61I and extending in some instances to MW-58S and/or MW-56D and MW-57S. For example, the highest levels of ethylbenzene (22 ug/L) and total xylenes (66 ug/L) were observed at MW-61I. The highest levels of other significant VOCs also were detected in the northeastern extent of the plume area: 1,1-DCA (detected in nine samples) and 1,1-DCE (detected in six samples) showed the greatest concentrations at MW56D (64 ug/L and 31 ug/L, respectively). The highest levels of 1,2,4-trimethylbenzene (seven samples up to 65 ug/L) and 1,3,4-trimethylbenzene (seven samples up to 18 ug/L) were detected at MW-61I and MW-55S, respectively.

As described in Sections 3.5.3 and 4.3.3, select FFTA wells were analyzed for perchlorate. There is no information that suggests that perchlorate is a contaminant associated with activities at the site. The analysis was included as part of a base-wide effort to respond to a national interest in defining the potential presence of perchlorate as a contaminant associated with military missile and rocket fuels. Five site wells and one background well were sampled and analyzed for perchlorate. Table 5.10 presents the perchlorate results for these wells. Perchlorate was not detected in any of the site wells or in a duplicate sample collected from MW-55S. It was reported as present at an estimated concentration of 3.4J ug/L (less than the detection limit of 4 ug/L) in the upgradient background well. The EPA considers perchlorate concentrations ranging between 4 ug/L and 18 ug/L as the target clean up goal for remedial actions.

5.5 REFINED CONCEPTUAL SITE MODEL (CSM)

The additional hydrogeological and chemical data obtained during the Supplemental RI were used to revise the CSM as presented in this section. The revised CSM for the FFTA was derived through the integration of the updated interpretation of groundwater flow with the nature and extent of contamination based on the results from the Supplemental RI sampling and select previous data.

The site is underlain by the Columbia Group and the Yorktown Formation, which comprise the two aquifers beneath the FFTA. The Columbia aquifer is the aquifer of concern at the FFTA because the underlying Yorktown aquifer is hydraulically isolated from the Columbia by a confining clay unit or aquitard. The

Columbia aquifer extends from the land surface to the top of the Yorktown aquitard, or to a depth of approximately 50 feet at the FFTA.

The Columbia Group consists chiefly of fine- to medium-grained sands, with lesser amounts of silt and clay. The hydrogeologic conclusions and the interpretation of the CSM for this current investigation differ somewhat from the previous investigations by the conclusion that the Columbia aquifer does not function as a homogeneous, interconnected aquifer. Rather, a silty clay layer that is approximately 3 feet thick and is consistently encountered at a subsurface elevation near sea level (referred to in this report as the Sea Level Clay, or SLC) has been interpreted to function as a leaky aquitard that hydraulically divides the Columbia aquifer into upper and lower units. Although some of the geologic data generated prior to the present study is incomplete, the cumulative data suggest that this clay may be continuous or nearly continuous beneath the FFTA. A similar clay unit has also been reported at or near the same elevation at the nearby Waste Oil Dump, suggesting this unit may have at least a semi-regional extent. One existing monitoring well at the FFTA (MW-56D) breaches and is screened across this clay.

Groundwater at the FFTA has been interpreted to flow in a predominantly north to northeastward direction, which is consistent with both the surface topography and the interpretations from previous investigations. Groundwater in the upper unit of the Columbia aquifer flows in a northeastward direction towards the unnamed tributary to Little Mosquito Creek and from there towards Little Mosquito Creek, which is the interpreted discharge point for most of the groundwater flowing from or through the FFTA. The shallowest groundwater (at and near the water table) may discharge into the unnamed tributary. Groundwater in the lower unit of the Columbia aquifer flows in a generally northward direction and does not appear to be influenced by the unnamed tributary, although the lesser number of observation points available for the lower unit make this interpretation somewhat less certain.

The subsurface soil data indicate that the primary source of groundwater contamination occurs in the immediate vicinity of soil borings SB-101, SB-104, and SB-105, which are located adjacent to each other in the area of the highest historical VOC soil gas detections and in the area of the former fire training pit. During the drilling of these three borings, elevated PID readings and organic odors were first detected at relatively shallow depths within the vadose zone (at depths of 3 to 7 feet), and these detections generally continued to the water table. The laboratory analytical data (Section 5.4) from these borings confirm that the soils are contaminated at depths well above the water table. The lack of significant PID readings or quantitative VOC laboratory detections from the soil borings drilled to the east, west, and south of these three borings supports their identification as being located within the historical source area. Elevated PID readings were also detected at depths just above the water table from the borings drilled immediately downgradient of the source area (SB-106 through SB-110), but the soils sampled from this horizon did not contain detectable VOCs, and significant PID readings were not detected at the shallower depths within

these borings. Therefore, these particular PID readings are interpreted to chiefly represent partitioning of groundwater volatiles into the vadose zone rather than significant soil contamination.

Conceptually, the vertical fluctuation of the water table typically creates a “smear zone” at and downgradient of the source area when some of the groundwater contaminants carried or introduced into the overlying soil during times of an elevated water table remain trapped as residual contamination in the vadose zone (or in the capillary fringe) with the subsequently falling water table. The lack of significant soil contamination immediately above the water table at the locations immediately downgradient of the source area apparently indicate that either a smear zone has not developed at this site, or that the water table was near its highest annual elevation at the time of sampling.

In summary, the subsurface soil data are interpreted to define a relatively discrete source area of contaminated soil in the vadose zone in the vicinity of soil borings SB-101, -104, and -105. This source area is bounded to the south, east, and west (the hydraulically upgradient and sidegradient directions) and to the north (the hydraulically downgradient direction) by uncontaminated soil. The elevated PID readings detected near the water table from the borings located in the downgradient direction (coupled with the lack of vadose zone contamination) are interpreted to reflect the significant groundwater contamination detected in this area rather than an extension of the shallower soil contamination.

The plume of contaminated groundwater at the FFTA formed chiefly through the downward percolation of meteoric water through the contaminated soil of the vadose zone. Conceptually, additional groundwater contamination may result from the migration (or “flow-through”) of clean, upgradient groundwater through a zone of residual (or immobile) groundwater contamination that often exists at and near a source area. The relatively low concentrations of organic contamination detected at the water table in MW-101S and MW-55S however, appear to indicate that a significant source of residual contamination does not currently exist below the water table at the FFTA.

No free product or non-aqueous contaminant phases (DNAPL or LNAPL) were detected in the investigation, nor is their presence suggested by the contaminant concentrations detected. The contaminants are interpreted to migrate from the source area and through the FFTA as a dissolved-phase plume within the site groundwater. Therefore, the plume is subject to the same hydraulic controls and influences as the site groundwater. The contaminant isoconcentration (plume) maps (Figures 5-4 through 5-7) and the hydrogeologic cross-sections (Figures 4-11 and 4-12) confirm that the migration pathways of the contaminant plume are similar to those interpreted for the groundwater as a whole.

The groundwater plume extends from the source area (near MW-101S and MW-55S) and migrates in a generally north and northeastward direction towards Little Mosquito Creek and towards the unnamed

tributary to Little Mosquito Creek. The effects of lateral dispersion appear to be significant (possibly reflecting or resulting from the low horizontal gradient) and create a plume extending from the area of well cluster MW-58S/MW-102D in the west to the unnamed tributary of Little Mosquito Creek to the east. Based on the analytical results from monitoring well MW-14GW-5 and the hydrogeologic cross-section A-A' (Figure 4-11), a portion of the plume is interpreted to discharge into the unnamed tributary.

The migration of the contaminant plume is essentially confined to the upper flow unit within the Columbia aquifer. The presence of the SLC appears to control the migration of the contaminants and influence the flow within the upper unit. To illustrate this, total VOC contaminant levels detected in the groundwater at FFTA wells have been plotted on the hydrogeologic cross sections and are shown in Figures 5-8 and 5-9. Although the extended screen intervals of MW-61I and MW-56D and the overlapping nature of the well screens at well cluster MW-55 complicate the interpretation, the groundwater plume is interpreted to migrate downward as it migrates away from the source area (see cross-section B-B', Figure 5-9), which is consistent with the downward vertical hydraulic gradients detected throughout the site. The SLC however, has been interpreted to significantly retard the downward migration of the plume as the groundwater contaminant concentrations detected from the wells screened below the SLC are consistently lower than the concentrations detected just above the SLC at the same locations (see Figures 5-8 and 5-9), including those located significantly downgradient from the source area (for example, well pairs MW-57S/MW-105D; MW-103I/MW-103D; and MW-58S and MW-102D). Therefore, the SLC appears to significantly control the vertical migration of the groundwater plume and retards contaminant migration to the deeper portion of the aquifer.

Based on the interpreted groundwater flow directions and the site topography, most of the FFTA groundwater plume would be expected to discharge in the vicinity of the confluence of the Little Mosquito Creek and its unnamed tributary. The monitoring well installed to test this hypothesis (MW-104S) did not contain any site contaminants, even though there has been more than sufficient time for the plume to migrate this lateral distance, given the age of the plume and the interpreted groundwater velocity within the Columbia aquifer. The absence of the plume at MW-104S is interpreted to indicate that the FFTA groundwater plume is attenuating at some point upgradient from this discharge zone.

A review of historical and current contaminant levels and measurements support the interpretation that attenuation and/or contaminant degradation is occurring within the FFTA plume. As presented in Section 5.4.3, the contaminant concentrations within the plume appear to be decreasing and the areal extent of the plume appears to be less extensive. Benzene, toluene, and cis-1,2-DCE have historically been used to define the extent of the groundwater contamination plume. Maximum benzene levels in the center of the plume (MW-61I) have decreased from 100 ug/L to 28 ug/L during the period from 1996 to 2003. Maximum historical toluene concentrations reported at 1,400 ug/L in MW-55S and 32 ug/L in MW-55D (Metcalf &

Eddy, Inc., 1996b; Versar, Inc., 2000) have decreased to 83 ug/L and non-detect in MW-55S and MW-55D, respectively. Similarly, current data suggest that the cis-1,2-DCE plume still exists in the same general area identified in historical data, but the contaminant levels in the center of the plume (MW-61I) have decreased from 3,000 ug/L in 1996 to 1,000 ug/L in 2000 and currently were detected at 460 ug/L.

These data and the plume data discussed in Section 5.4.3 suggest that the groundwater plume may be decreasing and also suggest the lack of a major continuing source of contamination. Various MNA data collected during this Supplemental RI including dissolved oxygen, ORP, and ferrous iron also support the interpretation that natural attenuation may be occurring within the plume. Table 4.6 presents field measurement data for each monitoring well collected during the Supplemental RI. Dissolved oxygen levels are markedly lower in wells MW-55S, MW-61I, MW-56D, and MW-103I than in other FFTA wells. Dissolved oxygen levels in these four wells ranged from 0.46 mg/L to 0.93 mg/L, compared to a range of 1.6 mg/L (MW-103D) to a high of 9.77 mg/L (MW-1) measured in the other FFTA wells. These four wells are located along the approximate center line of the contaminant plume, and the depressed oxygen levels are consistent with degradation processes. Similarly, the lowest ORP measurements, -96 mv and -45 mv, and highest ferrous iron concentrations, 3.6 mg/L and 9.5 mg/L, were identified in monitoring wells MW-55S and MW-61I, respectively. These data, coupled with the groundwater contaminant data, suggest that the magnitude and extent of the plume may be decreasing through natural attenuation and degradation processes.

6.0 FATE AND TRANSPORT

Several aspects of contaminant fate and transport relevant to the FFTA are discussed in this section. The ultimate fate of chemicals in the environment is determined by several physical, chemical, and/or biologically related factors. The role that physical properties such as specific gravity, solubility, and vapor pressure play in determining the processes that take place for a particular chemical (or class of chemicals) is often variable and can vary considerably from location to location, even within the same contaminant regime. Chemical and biological transformational processes can also be significantly affected by localized conditions.

Various chemical and physical properties affecting contaminant migration and the media (e.g., geologic influences) that affect their migration are discussed in Section 6.1. No distinctions as to the locations or concentrations of the chemicals detected at or near the site are made in this section. Section 6.2 presents a discussion of contaminant persistence. Potential contaminant migration routes are identified and discussed in Section 6.3. Section 6.4 summarizes fate and transport mechanisms as they pertain to the FFTA.

6.1 PHYSICAL AND CHEMICAL TRANSPORT PROPERTIES

This section provides a qualitative discussion of potential migration of the contaminants found at the FFTA. The physical and chemical properties of the chemicals found in the study area, where available, are presented in Table 6.1. Partitioning coefficient ranges for detected metals are presented in Table 6.2. These parameters may be used to assess the behavior of a chemical in the environment.

Empirically determined literature values of water solubility, octanol/water partition coefficient (K_{ow}), organic carbon partition coefficient (K_{oc}), soil-water partitioning coefficient, vapor pressure, Henry's law constant, and specific gravity are presented for organic chemicals, as available. Many of these parameters are not applicable to inorganic chemicals. For inorganics, specific gravity has been presented along with qualitative descriptions of important environmental fate properties. Calculated values, which were obtained using approximation methods, are presented when literature values are unavailable. A discussion of the environmental significance of each of these parameters follows.

6.1.1 Solubility

The rate at which a chemical is leached from a waste deposit by infiltrating precipitation is proportional to its water solubility. Chemicals that are more soluble are expected to enter water much more readily and rapidly than less soluble chemicals. The VOCs (ketones, monocyclic aromatics, and halogenated

aliphatics) are usually several orders of magnitude more water soluble than the PAHs, PCBs, and chlorinated dioxins/furans.

At the FFTA, metals were detected in both soil and groundwater. When detected in groundwater samples, they may be part of the suspended sediment in the sample or dissolved in the aqueous fraction.

The solubility of inorganics is strongly influenced by their valence states and forms (e.g., hydroxides, oxides, carbonates). The solubility is also dependent on pH, Eh, and the presence of other ionic species in solution (the Debye-Huckel theory). Solubility factors reported in the literature vary with the type of complex formed.

In the case of manganese (Mn), which is a common metal and occurs in several natural mineral forms, the predominant species is Mn^{+2} as the solvated (or hydrated) ion at normal pH and ORP values. This ionic state is typically associated with metamorphic rocks. Other manganese compounds, such as permanganate (Mn^{+7}) or manganese oxide (MnO_4), are either too easily reduced to be present in ambient waters or have low solubilities.

6.1.2 Octanol-Water Partitioning Coefficient (K_{ow})

The K_{ow} is a measure of the equilibrium partitioning of a chemical between octanol and water. The K_{ow} is also used to estimate bioconcentration factors (BCFs) in aquatic organisms. A linear relationship between the K_{ow} and the uptake of chemicals by fatty tissues of animal and human receptors has been determined (Lyman et al., 1990). PAHs, phthalate esters, PCBs, and chlorinated dioxins/furans are several orders of magnitude more likely to partition to fatty tissues than the more water-soluble VOCs. The K_{ow} is also useful in characterizing the sorption of compounds by organic soils when experimental values are not available. Relatively simple organic chemical molecules have low K_{ow} values.

6.1.3 Organic Carbon Partitioning Coefficient

The soil/sediment partition (organic carbon partition) coefficient (K_{oc}) is related to water solubility and K_{ow} . K_{oc} indicates the tendency of a chemical to bind to soil particles containing organic carbon. Chemicals with high K_{oc} generally have low water solubilities and vice versa. Chemicals such as semivolatiles, PAHs, PCBs, and chlorinated dioxins/furans are relatively immobile in the subsurface environment and are preferentially bound to the soil phase. These compounds are not subject to groundwater transport to the same extent as compounds with high water solubilities.

K_{oc} may be used to infer the relative rates at which are transported in groundwater. Complex organic chemicals are relatively immobile and are preferentially bound to the soil phase. These compounds are not subject to rapid groundwater transport.

6.1.4 Distribution Coefficient

The soil-water partitioning (distribution) coefficient (K_d) is a measure of the equilibrium distribution of a chemical or ion in soil/water systems. The distribution of organic chemicals is a function of both the K_{oc} and the fractional organic carbon content of the soil (FOC) for the solid and aqueous matrices.

6.1.5 Vapor Pressure

Vapor pressure provides an indication of the rate at which a chemical volatilizes from both soil and water. It is of primary significance in instances where environmental interfaces such as surface soil/air, sediment/air, and surface water/air are important, rather than in evaluation of groundwater and subsurface soils. Vapor pressures for VOCs are generally many times higher than vapor pressures for other organic compounds (e.g., semivolatiles, pesticides, and PCBs). Chemicals with higher vapor pressures are expected to enter the atmosphere much more readily than chemicals with lower vapor pressures. Volatilization depends upon such factors as total exposed surface area, contaminant concentrations, and amount of organic matter in the soil.

6.1.6 Henry's Law Constant

Both vapor pressure and water solubility are of use in determining volatilization rates from surface water bodies and groundwater. The Henry's Law constant (HLC) is the equilibrium vapor pressure of a chemical above a solution divided by its concentration in the solution (for dilute systems). The HLC may also be used to calculate the equilibrium contaminant concentrations in the vapor versus liquid phases for dilute solutions commonly encountered in environmental settings. The HLC is also useful for mass transfer applications for air-stripping column design.

In general, chemicals with a HLC less than 1×10^{-5} atm·m³/mole should volatilize very little and be present only in minute amounts in the atmosphere or in soil gas. Chemicals with a HLC greater than 5×10^{-3} atm·m³/mole, such as halogenated aliphatics and monocyclic aromatics can undergo significant volatilization and diffusion in soil gas. HLCs (generally measured under laboratory conditions) for some of the PAHs, semivolatiles, pesticides, and PCBs indicate that these chemicals would tend to volatilize at a limited extent. However, in the environment these chemicals are strongly sorbed to soil/sediment particles that would keep them from volatilizing into the atmosphere.

6.1.7 Specific Gravity

Specific gravity is the ratio of the weight 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 pure compounds or very high concentrations of a contaminant will float or sink in water.

Chemicals with a specific gravity less than 1.0 will tend to float, whereas chemicals with a value greater than 1.0 will tend to sink. Of the commonly detected chemicals at the FFTA, the ketones and monocyclic aromatics have a specific gravity less than 1. Halogenated aliphatics, PAHs, PCBs, phthalate and esters have a specific gravity greater than 1.

6.2 CONTAMINANT PERSISTENCE

The persistence of the classes of organic and inorganic contaminants found in the study area is discussed in this section. Several transformation processes are believed to affect the persistence of organic chemicals in the environment. The primary processes affecting contaminant fate for organic compounds include microbial, photolytic, chemical degradation (hydrolysis, biodegradation), and oxidation/reduction reactions. For inorganics, persistence may be influenced by soil characteristics such as cation exchange capacity (CEC), TOC, pH, and soil porosity. General classes of the detected chemicals are addressed because the fate of chemicals in the environment is usually similar for chemicals within a particular chemical family.

The following general classes of contaminants are discussed; all of these classes have been detected in samples from the site:

- Monocyclic aromatics
- Halogenated aliphatics
- PAHs
- Phthalate esters
- Metals
- PCBs
- Chlorinated dioxins/furans

Table 6.3 summarizes those contaminants that were grouped into classes for evaluating persistence. These contaminants include inorganics and organic compounds that were detected at the site (see Section 5.0).

In general, photolytic degradation is not considered a relevant degradation mechanism for monocyclic aromatic compounds such as toluene (EPA, 1982). Furthermore, even photodegradable compounds must be present in media exposed to sunlight for such degradation to occur (such as surface soil). Because only a few VOCs were detected in surface soil at the FFTA, photolytic degradation is not expected.

Generally, organic molecules are subject to several chemical reactions under environmental conditions. Such reaction mechanisms include acid/base reaction, addition, elimination, and hydrolysis. However, monocyclic aromatics are not particularly amenable to the majority of these degradation mechanisms. Hydrolysis is considered to be negligible for halogenated hydrocarbons. Hydrolysis reactions can occur under acidic, basic, or neutral conditions.

Another possible type of degradation is the dechlorination of tetrachloroethene and trichloroethene to 1,2-DCE, 1,1-DCE, and ultimately vinyl chloride (Cline and Viste, 1984). A similar type of reaction has been reported for 1,1,1-TCA in degradation to 1,2-DCA, 1,1-DCA, and chloroethane. Because some of these compounds (TCA, 1,2-DCE, vinyl chloride) were detected in the groundwater samples collected at the site, these reactions may be occurring in the natural environment. The presence of high levels of organic carbon would encourage degradation. Therefore, these processes may be occurring at the site.

Bacterial degradation is a potential environmental fate mechanism. A compound for which biodegradation is potentially considerable is toluene (EPA, 1982; Verschueren, 1983). This degradation is primarily biological, and anaerobic conditions are typically required for these reactions to occur.

6.2.1 Monocyclic Aromatics

Monocyclic aromatics, such as benzene, toluene, 1,2,4-trimethylbenzene, and xylenes (BTEX compounds), are not considered to be persistent environmental contaminants in comparison to pesticides, PCBs, phthalate esters, and metals. Such compounds are subject to degradation via the action of both soil and aquatic microorganisms. The biodegradation of these compounds in the soil matrix is dependent on several factors including the abundance of microflora, macronutrient availability, soil reaction (pH), temperature, and oxygen.

Although BTEX compounds are amenable to microbial degradation, the rate of degradation cannot be predicted without information on the availability of nutrients and the type of bacteria present. If these contaminants discharge to nearby streams, volatilization and biodegradation may occur relatively rapidly. For example, a reported first-order degradation rate constant for benzene is 0.11 day^{-1} in aquatic systems (Lyman et al., 1990). This corresponds to an aquatic half-life of approximately 6 days.

Additional degradation processes such as hydrolysis and photolysis are considered to be insignificant fate mechanisms for monocyclic aromatics (EPA, 1982). However, some monocyclic aromatic compounds such as benzene and toluene have been shown to undergo clay-, mineral-, and soil-catalyzed oxidation (Dragun, 1988).

6.2.2 Halogenated Aliphatics

Halogenated aliphatics, such as 1,2-DCE and 1,1,-TCA, are subject to reductive dehalogenation via the action of anaerobic bacteria. Research indicates that degradation of highly chlorinated ethanes is a relatively slow process. It does not appear that appreciable degradation of halogenated aliphatics occurs in aerobic aquatic systems (EPA, 1982) or unsaturated soils (Lyman et al., 1990). Photolysis is not considered to be a relevant degradation mechanism for this class of compounds (EPA, 1982). Limited hydrolysis of saturated aliphatics (i.e., alkanes) may occur, but it does not appear to be a significant degradation mechanism for unsaturated species (i.e., alkenes) (EPA, 1982).

Transport of halogenated aliphatics from groundwater to surface water can occur. When discharged to surface water, these compounds tend to volatilize rapidly due to their high solubility rather than bind with sediments.

6.2.3 PAHs

PAHs are common constituents of oil and grease and may be present at the site due to incomplete combustion of burned materials at the site. Data from landspreading applications have indicated that PAHs are amenable to microbial degradation. Studies have demonstrated that PAHs are much more amenable to degradation in soil matrices than in aquatic environments (EPA, 1979). Under existing site conditions, the rate of microbial degradation cannot be predicted without knowledge of microbial populations. PAHs do not contain functional groups that are susceptible to hydrolytic actions, and hydrolysis is considered to be an insignificant degradation mechanism. Photolysis may be a major degradation mechanism in aquatic environments but is probably insignificant in solid materials.

6.2.4 Phthalate Esters

Phthalate esters are considered to be relatively persistent environmental contaminants. Although numerous studies have demonstrated that phthalate esters undergo biodegradation, it appears that this is a very slow process in both soil and surface water. Certain microorganisms have been shown to excrete products that increase the solubility of phthalate esters and enhance their biodegradation (Gibbons and Alexander, 1989). Biodegradation of bis(2-ethylhexyl) phthalate and other phthalate esters is an important fate mechanism, as

is bioaccumulation. Hydrolysis of phthalate esters is very slow, with calculated half-lives of 3 years (dimethyl phthalate) to 2,000 years [bis(2-ethylhexyl) phthalate] (EPA, 1979). Similarly, photolysis is considered to be an insignificant degradation mechanism for phthalate esters (EPA, 1982).

6.2.5 PCBs

PCBs are considered to be very persistent organic chemicals. Biodegradation is the only process known to transform PCBs under environmental conditions, and only the lighter compounds are measurably biodegraded (EPA, 1979). Although some microorganisms may biodegrade PCBs, such fungi may not exist in local soil. There is experimental evidence to suggest that heavier PCBs (five or more chlorines per molecule) can undergo photolytic degradation, but there are no data to suggest that this process operates under environmental conditions (EPA, 1979). Base-, acid-, and neutral-promoted hydrolysis are considered to be inconsequential degradation mechanisms for PCBs (EPA, 1982).

6.2.6 Chlorinated Dioxins/Furans

Polychlorinated dioxins/furans (PCDDs/PCDFs), reported as 2,3,7,8-TCDD toxicity equivalents, are usually released to the environment primarily through emissions from the incineration of municipal and chemical wastes, in exhaust from automobiles using leaded gasoline, and from the improper disposal of certain chlorinated chemical wastes. If released to the atmosphere, vapor-phase PCDDs/PCDFs may be degraded by reaction with hydroxyl radicals and direct photolysis. Particulate-phase PCDDs/PCDFs may be physically removed from air by wet and dry deposition. If released to soil, PCDDs/PCDFs are not expected to leach. Photo degradation on terrestrial surfaces may be an important transformation process. Volatilization from soil surfaces during warm conditions may be a major removal mechanism. The persistence half-life of TCDD on soil surfaces may vary from less than 1 year to 3 years, but half-lives in soil interiors may be as long as 12 years. Screening studies have shown that TCDD is generally resistant to biodegradation. The major route of exposure to the general population results from incineration processes and exhausts from leaded gasoline engines (TOXNET, online, November 2001).

6.2.7 Metals

The transport and fate of metals in the environment are primarily controlled by sorption to soil/sediment material. Some key parameters affecting mobility of metals in groundwater are listed in Table 6.2. The metal-organic relationships, both in soil and water, increase in importance as the organic carbon content increases. Some metals (e.g., arsenic) are extremely soluble and mobile in the environment. Many other metals, such as zinc and copper, have an affinity for hydrous iron and manganese oxides, as well as for

organic materials, and are therefore preferentially adsorbed to soil. The mobility of most metals increases as the soil pH decreases. Table 6-3 presents the metals of concern for the various media.

Inorganic contaminants with a positive charge (cations) will be retarded by clays exhibiting a net negative charge, and anions such as chromium (as chromate) and arsenic (as arsenate) will be more mobile in such an environment. The mobility of metals in soil is also influenced by the metals' ability to form insoluble precipitates, bind to metal oxides in the soil, or remain insoluble.

6.2.8 Summary

A summary of the fate processes for each of the critical compounds found in the study area is shown in Table 6.4. This table helps identify which compounds would be the most persistent in the environment. Metals and PAHs are the most persistent compounds found that may be attributable to the site. The metals detected do not undergo volatilization, biodegradation, or photolysis as fate processes.

6.3 CONTAMINANT MIGRATION FACTORS AND ROUTES

In general, several potential migration routes exist in areas contaminated with hazardous materials. For the FFTA, such migration routes include, but are not limited to,

- Atmospheric migration via particulate or volatile vaporization or fugitive dust emissions from the site
- Overland migration of dissolved or adsorbed contaminants
- Leaching from the site to underlying groundwater
- Groundwater transport from the site

Contaminants migrating from the site may be transported through one or more environmental media. The physical and chemical characteristics of those media affect the interactions between the contaminant and the media and determine how, or if, a given contaminant migrates in a particular medium. The environmental media identified in Section 5.0 as being potentially contaminated by site activities include surface and subsurface soil and groundwater.

The environmental factors that have the greatest influence over the mobility of contaminants in the various media include

- Hydraulic conductivity
- Cation exchange capacity (CEC)
- Organic carbon content

- Hydraulic Gradient
- pH
- Oxidation /reduction potential
- Groundwater/surface-water interactions
- Dissolved oxygen
- Temperature
- Specific conductance

6.3.1 Atmospheric Migration

The first possible route for contaminant migration from the site through the air is the volatilization of compounds from surface water or surface and subsurface soil (especially if exposed through excavation or erosion). The results of the sampling and analysis indicate that this is not happening on a significant scale on a site-wide basis. VOCs were infrequently detected and at low (near quantitation limits) levels in surface soils.

The second possible route for contaminant migration from the site through the air is the generation of contaminant dusts from surface soil. This would be potentially important for metals and possibly for organic chemicals strongly sorbed to soil particles. Subsurface soil fugitive dust exposure would depend upon the subsurface material being exposed in some way. Potential future risks associated with estimated dust emissions via inhalation of suspended particles from soils are assessed in Section 7.0.

6.3.2 Soil Migration

Surface soils may be transported by water erosion during rainstorms and other precipitation events. Along with overland flow of contaminants, wind erosion may be responsible for the movement of fine-grained particles on site. The soil properties that affect permeability or potential reaction with the dissolved components in the water include not only permeability and hydraulic conductivity but also CEC, mineral content of the soil, and the soil organic carbon content. Overburden materials present at the site are predominantly sandy loams.

Samples from the FFTA indicate that substantial levels of organic contamination of surface soils are not present at the FFTA. Metals are present but at levels similar to background levels. The migration of contaminants through the soil partly depends on the capacity for fluid movement through the subsurface. Hydraulic conductivity is a measure of the rate at which water flows through the soil or rock under a change in pressure or head. Soils generally classified as sandy loams are considered well-drained.

The fate of metals in surface soils may be largely controlled by weathering and the absorptive capacity of the soil. Absorption will control the amount of metals that can be retained within the soil. The capacity of soil to immobilize ionic inorganics can be reasonably predicted based on a correlation equation involving pH and CEC. CEC is a measure of the capacity of the soil to adsorb positively charged ions such as metals onto the negatively charged surface of the soil particles. Although soils were not analyzed for CEC during the FFTA investigations, it is likely that they would exhibit high CEC values and would have a generally high capacity to adsorb metals.

Generally, a soil composed of clay or a clay mixture attenuates (slows) dissolved ions during transport more than does a clay-free soil. Conversely, a soil composed of well-sorted (i.e., clay-free) quartz sand and gravel typically has little or no capacity to adsorb or exchange an ionic substance (e.g., metals or radionuclides) migrating through it. Therefore, soils present at the site are not expected to significantly attenuate contaminants via this mechanism as they consist of well-drained sandy loams.

The amount of organic carbon present in a soil also increases the relative amount of an organic compound that will be attenuated by that soil. In effect, the organic carbon content increases the effective K_d of chemical species in solution. The greater the organic carbon content in the soil, the more likely it is that organic compounds migrating through the soil will become adsorbed onto the soil and the less likely they will migrate long distances through it. If appropriate pathways exist, contaminants retained on soils or sediments could migrate off site with surface water flowing away from the site. However, as described in earlier sections, the FFTA site is surrounded by areas of higher elevation, and surface waters infiltrate or evaporate from the site rather than flow by overland route to surface water bodies.

The soil pH helps predict the mobility of inorganic species and is defined as a measure of the hydrogen ion concentration. The soil pH is important with respect to a metal's ability to remain soluble or to precipitate as a particular salt. Several metals, including lead, are soluble at both basic and acid pH conditions. These metals are considered amphoteric. The soil at the FFTA is classified as acidic in nature as pH ranges from 5.0 to 7.0.

6.3.3 Groundwater Migration

Migration and transport of contaminants in groundwater are release mechanisms at the site. Both organic and inorganic contaminants from the surface and subsurface soils can enter the groundwater and be transported vertically and horizontally. The FFTA groundwater flow patterns are detailed in Section 4.5. Groundwater vertical gradients are generally in a downward direction, and groundwater flow is influenced by the presence of a clay lens. Flow in the upper groundwater zone, above the clay lens, is to the east and northeast and tends to mimic the local topography. The lower flow zone is to the north. Both flow

zones are considered groundwater recharge zones at the FFTA. Little Mosquito Creek, located to the north of the FFTA, is expected to be the regional groundwater discharge zone.

The groundwater analytical data from the site revealed the presence of VOCs. Advective transport and mechanical dispersion in groundwater are the principal mechanisms by which VOCs and other contaminants are transported away from the site.

The first and most likely route of contaminant migration is through surface soil to subsurface soil, where contaminants enter the upper flow zone. As water infiltrates through a contaminated zone, partitioning from solid to aqueous phase will occur. The amount of chemical that will dissolve into infiltrating water is determined by a number of factors including residence time, solubility, partitioning factor, and pH of the water. The dissolved chemicals continue downward and are able to interact with stationary (soil) particles in the saturated and unsaturated zones. After percolating through the capillary zone, dissolved contaminants are then able to enter groundwater, where transport can occur in bulk through advection. The chemical concentrations in groundwater increase significantly shortly after initial groundwater impact to a maximum level. The longer-term effects at the source are a gradual decrease in the concentrations over time as chemical removal from the source area occurs. Short-term variations in release rate and impact to groundwater can occur, but long-term trends of decreased levels are usually observed. Diffusion and dispersion throughout the groundwater flow regime occur as the groundwater flows.

During groundwater contaminant transport, a number of processes help to reduce chemical concentrations. Diffusion and attenuation effects are non-transformational mechanisms that result in a direct decrease in chemical concentrations. Chemical and biological reactions with dissolved chemicals can also result in decreased concentrations. The products of these reactions, however, may have significantly different chemical, transport, and toxicological properties from the parent compounds.

Sampling results indicate that VOCs have migrated horizontally in groundwater.

6.4 SITE-SPECIFIC ENVIRONMENTAL FATE OF CONTAMINANTS OF POTENTIAL CONCERN

This section identifies the potential contaminant release and transport mechanisms identified through evaluation of the chemical analytical data and known characteristics of the site. This section will also address existing and potential migration routes at the site, in particular migration through surface and subsurface soil and groundwater.

Three basic factors influence the potential migration of contaminants:

- Physical and chemical properties of the specific compounds
- Physical and chemical characteristics at the site and in the surrounding study area
- Volume and concentration of contaminants.

6.4.1 Surface and Subsurface Soils

As evidenced by the analytical data, metals and various organic compounds including monocyclic aromatics, halogenated aliphatics, PAHs, and VOCs were detected at generally low levels in surface and subsurface soil at the site. The frequencies of detection was also low and confined to a small area around the location of the former fire training pit.

The VOCs detected at the site are highly mobile in soil environments, and would tend to move downward into the groundwater column. The other organics and metals detected in the FFTA soils are generally highly sorbed onto soil particles and generally immobile in the environment. Site data indicate that VOCs are present at the highest concentrations and frequencies in the subsurface area around the location of the former fire training pit and that the levels are relatively low. The contaminant pattern present in these soils is similar to the contaminant pattern seen in downgradient groundwater samples, indicating that the subsurface soils may be associated with the source of the groundwater contamination. However, the level of contaminants present in the subsurface soils (see Section 5) is considered relatively low in comparison to the groundwater levels. In addition, the contaminant concentrations detected in the subsurface soils and groundwater are below the EPA soil screening guidance for the soil to groundwater pathway. The contaminant levels detected in FFTA soils were qualitatively reviewed to evaluate the soil to groundwater pathway. Table 6.5 presents the results of this evaluation. The EPA screening criteria is shown for those contaminants detected in site groundwater and site soils and for metals, were present at levels greater than background. For each substance, maximum concentrations found in soils were compared to their respective soil screening level (SSL) (EPA, 1996c). EPA's online SSL calculator was used to define the SSLs. Assumptions for the SSLs are shown in the footnotes to Table 6.5 and in Appendix G. No groundwater contaminant was detected in site soils at levels that exceed background and the screening criteria.

The site data suggests that FFTA soils are not currently a significant source of contaminant release to the environment. Past removal actions and/or years of exposure to infiltration of precipitation and groundwater flushing may have removed the majority of soluble contaminants from the soils.

6.4.2 Groundwater

As evidenced by analytical data, the major groundwater contaminants detected at the site in the surrounding study area were volatile organics, specifically halogenated aliphatics and monocyclic aromatics. It is believed

that these compounds were introduced at the former fire training pit and migrated from their original location. Precipitation, a natural flushing agent, dissolved soluble components and carried them down to the water table. After reaching the groundwater, the waste components were transported downgradient and currently are present primarily in the form a groundwater plume contained within the upper flow zone. Section 5 presents a detailed evaluation of the nature and extent of VOC contamination in groundwater and presents a detailed conceptual model for the FFTA.

Contaminant mobility is dependent on characteristics such as solubility and partitioning ability based on partition coefficients. The VOC contaminants detected in groundwater have high solubilities and low K_{oc} values making them highly susceptible to groundwater transport. The halogenated aliphatic degradation processes, that include PCE, 1,1,1-TCA, 1,1-DCA, 1,1-DCE, 1,2-DCE and vinyl chloride may be occurring in groundwater beneath the site. In addition, field measurements of dissolved oxygen, ORP and ferrous iron indicate that biodegradation of other compounds may also be occurring. As detailed in Section 5.5, low dissolved oxygen and ORP levels and high ferrous iron levels exist in the center of the groundwater plume. These measurements are indicative of a reducing environment and indicate that an anaerobic condition exists within this area. Historical and current analytical data also suggest that the magnitude and extent of the plume is decreasing. The metals detected in groundwater may also be related to the reducing environment and may be indicative of dissolved metals being released from native materials.

7.0 HUMAN HEALTH RISK ASSESSMENT

This section provides a description of the risk assessment methods employed for the supplemental soil and groundwater risk assessment for the NASA WFF Former Fire Training Area (FFTA), as well as a summary of the soil and groundwater risk assessment results. The general objectives of the risk assessment were to estimate the actual or potential risks to human health resulting from the presence of contamination in area groundwater, surface soil, and disturbed soil (footnoted as soil* to denote surface and subsurface soils that have been mixed), and to provide the basis for determining appropriate remedial measures (if necessary) for these media as part of the Feasibility Study.

7.1 INTRODUCTION

The specific objectives of the risk assessment were as follows:

- To estimate the actual or potential risks to human health resulting from the presence of contamination in area groundwater, surface soil, and soil* (mixture of surface and subsurface soils), at designated areas/media of concern.
- To provide a basis for establishing concentrations that are protective of potential human receptors under industrial, construction, and residential exposure scenarios.
- To determine the need for remedial measures (if applicable) for these media.

Three major aspects of chemical contamination must be considered when assessing public health risks: (1) contaminants with toxic characteristics must be found in environmental media and must be released by either natural processes or by human action; (2) potential exposure points must exist either at the source or via migration pathways if exposure occurs at a remote location other than the source; and (3) human or environmental receptors must be present at the point of exposure. Risk is a function of both toxicity and exposure; without any one of the three factors listed above, there is no risk.

This risk assessment is divided into Data Evaluation, Estimation of Exposure Point Concentrations, Exposure Assessment, Toxicity Assessment, Risk Characterization, Uncertainty Analysis, and Summary/Conclusions. Each section is briefly discussed below. Note that the tables associated with this section follow a standardized format adopted by EPA Risk Assessment Guidance for Superfund (RAGS), Volume I, Part D: Standardized Planning, Reporting, and Review of Superfund Risk Assessments.

Several aspects of Data Evaluation (Section 7.2) are primarily concerned with analytical data treatment methodology and were discussed in the preceding section, including handling of qualifiers and nondetects, averaging of duplicates, distributional analysis of the data, and background comparison tests. Once these steps are complete, chemicals of potential concern (COPCs) are selected for each medium that represent the potential contributors to significant site-associated risks. COPCs for each medium of interest are summarized on RAGS D Table 2s.

Estimation of Exposure Point Concentrations (Section 7.3) provides a statistical procedure for estimating the chemical input into each of the exposure pathways. The shape of the data distribution and magnitude of the standard deviation for each COPC define the basis for calculating the upper 95 percent confidence limit on the mean concentration, which is the input concentration of a chemical used to estimate the site-associated risks. Different methods of calculating EPCs apply for lead. EPCs are presented for each medium of interest on RAGS D Table 3s.

Exposure Assessment (Section 7.4) identifies categories of potential human health exposure based upon a characterization of the site setting, delineates potential receptors consistent with current and possible future land use patterns, outlines possible exposure routes by medium, documents the parameters and equations used in exposure estimates for each pathway, and explains the exposure evaluation used in blood-lead modeling. Exposure input parameters for each receptor and exposure pathway are presented on RAGS D Table 4s.

Toxicity Assessment (Section 7.5) presents available reference doses, cancer slope factors, EPA weight of evidence, adjustment of the dose-response parameters, chemical-specific dermal absorption factors, and toxicity equivalency factors (TEFs) for dioxins/furans. In this section, toxicity factors for noncancer toxicity and cancer risks are presented for all COPCs on RAGS D Table 5s and 6s, respectively.

Risk Characterization (Section 7.6) presents the approaches for determining carcinogenic risks, noncarcinogenic risks, and lead risks. The risk characterization quantitatively evaluates the potential for adverse health effects from exposure to COPC concentrations in environmental media by integrating information developed during the toxicity and exposure assessments. In this section, RAGS D Table 9s present noncancer risks subdivided according to target organ and a total of cancer and noncancer risks summed across all COPCs for each environmental medium, exposure pathway, and receptor. An overall cumulative summary of risks across all media and pathways of exposure is also presented for each receptor.

Uncertainty Analysis (Section 7.7) is a discussion of the general and site-specific uncertainties associated with the estimated risks, exposure models, and assumptions utilized in the HHRA.

7.2 DATA EVALUATION

7.2.1 Identification of COPCs

The selection of COPCs was based on chemical-specific concentrations, occurrence, distribution, and toxicity. Groundwater, surface soil, and subsurface soil analytical data were used to select COPCs by examining only those chemicals with positive detections. Groundwater and surface soil data were evaluated as separate media to which receptors might be exposed, whereas data from surface and subsurface soils were combined to allow evaluation as a single exposure medium (denoted as soil*) that represents potentially disturbed soil for future potential receptors. All site-related subsurface soil samples were evaluated in Sections 5 and 6 for the purpose of evaluating the overall extent of contamination and for soil-to-groundwater pathway screening. However, subsurface soil samples collected from depths below 15 feet are not considered available for potential exposure and are not relevant to the human health risk assessment. While future site activities may result in mixing and redistribution of subsurface soils, the EPA guidance generally limits the potential exposure depth to 10 feet and VADEQ guidance limits the depth of consideration to 15 feet or to groundwater, whichever is encountered first. Six FFTA subsurface soil samples were collected from depths greater than 15 feet. For human health risk assessment purposes, these samples were omitted from the data.

A substance was considered a candidate COPC for an exposure medium if the maximum detected concentration was greater than the associated risk-based concentration (RBC) based on a target cancer risk of 1×10^{-6} or a noncancer hazard quotient (HQ) of 0.1. RBCs were obtained from the latest EPA Region 3 Risk Based Concentrations (RBCs) listing for soil and tap water (EPA, 2003a). All exposures to groundwater, surface soil, or soil* for all receptors were conservatively screened using the residential exposure assumptions in the RBC table. RBCs that were based on noncancer effects were adjusted from a HQ of 1.0 to a HQ of 0.1 to protect against the possibility of additive toxic effects from multiple chemicals.

For soil media, candidate COPC metals were eliminated from quantitative risk calculations if none of the statistical tests indicated that levels were greater than background. [Such metals might contribute to background risks but not to site-related risks and are discussed further in the Uncertainty Analysis (Section 7.7)] The statistical methodology for comparing site data to background was presented in section 5.3. Results of background comparisons for surface soil are discussed in Section 5.3.4 and are presented in Table 5.5. For subsurface soil, relevant background comparison test results are presented in Appendix G. (Note that the table of background comparison results shown in Appendix G is considered relevant only to the direct contact exposure evaluation for the human health risk assessment because it omits soil samples collected at depths below 15 feet, while six additional samples from depths

below 15 feet were added to the data set used for the purpose of evaluating the overall site-related extent of contamination and for soil-to-groundwater pathway screening in Section 5.)

For surface soil, metals that exceeded RBCs were appropriately considered as COPCs for quantitative risk calculations if the outcome of any quantitative statistical test was above background (Y) or if all background tests were considered as not applicable (NA) because of diminished or unreliable capacity to detect a significant difference between site and background. (This can occur, for example, if there are very few samples, too many nondetected results, or other problems with the data.) For the evaluation of candidate COPC metals in soil* (mixture of surface and subsurface soil) two separate series of background tests - one for surface soil and one for subsurface soil - were considered in conjunction. Candidate COPC metals in soil* were eliminated from the risk assessment based upon background tests if at least one of the two data sets (either surface or subsurface soil) was qualified as not above background (N) and neither data set was judged to be above background (Y) for a particular COPC.

COPCs for metals present in groundwater were selected based only upon RBC comparisons and in lieu of any statistical background comparisons. Background tests were not considered reliable for groundwater because of a statistically inadequate number of site-specific upgradient sample locations (data were collected only for two upgradient wells). However, a qualitative comparison of site and background samples for groundwater is presented within the Nature and Extent of Contamination (Section 5).

Other considerations were applied to evaluate whether chemicals that do not have published RBC criteria should be selected as COPCs. Essential nutrients were not considered as COPCs, including calcium, chloride, magnesium, potassium, and sodium. Where possible, detected chemicals that did not have published toxicity criteria were compared to surrogate RBC criteria that were adopted from available RBCs for substances having similar chemical structure. The RBC for fluoranthene was used as a surrogate for evaluating phenanthrene and benzo(g,h,i)perylene since the latter two substances do not have published RBCs. Other detected chemicals that did not have published toxicity criteria from accepted references (EPA, IRIS, HEAST, or NCEA) were not retained as COPCs but are addressed in the Uncertainty Analysis (Section 7.7) to document cases where a current lack of knowledge regarding toxicity adds uncertainty to the risk assessment.

A lead value of 400 mg/kg [Office of Solid Waste and Emergency Response (OSWER) Directive, EPA, 1994a] was used as the residential soil screening RBC level and was applied to surface soil and soil* (mixture of surface and subsurface soils). Lead was evaluated as a potential COPC in groundwater based upon comparison to the MCL of 15 ug/L.

The RBC for hexavalent chromium was used for COPC selection because speciation data (i.e., trivalent versus hexavalent) were not available for samples collected at FFTA. Similarly, the RBC for methyl mercury was used for COPC selection because the form of mercury at the site is unknown and methyl mercury is considered the most toxic form of mercury.

Results of selection of COPCs for each of the media of concern at FFTA are presented in the following subsections. The COPC selections and the rationale for inclusion or exclusion of chemicals are presented in Tables 7.1 through 7.3. Chemicals with a "Y" listed in the COPC selection column of each table were retained as COPCs for all quantitative risk calculations.

7.2.1.1 Surface Soil COPC Selection

The following substances were selected as COPCs in surface soil and are shown on Table 7.1:

- Total 2,3,7,8-TCDD Toxicity Equivalents
- Aroclor-1260
- Benz(a)anthracene
- Benzo(a)pyrene
- Benzo(b)fluoranthene
- Dibenz(a,h)anthracene
- Indeno(1,2,3-cd)pyrene

While the above five PAHs exhibited maximum concentrations above screening criteria, twelve additional PAHs were detected at levels below RBCs and therefore are not expected to contribute to significant site-related risks. Eight VOCs were detected at concentrations less than RBCs and at low frequencies of detection (from one to five samples). Maximum concentrations of the metals aluminum, arsenic, and iron exceeded their respective RBCs but all of these metals were found to be not above background based upon statistical tests shown in Table 5.5.

While COPC selection was contingent upon maximum detected concentrations exceeding RBCs, from a risk-based perspective it is significant to note other descriptive statistics to better understand the range of detected concentrations that may affect risk for specific COPCs. For example, benzo(a)pyrene and dibenz(a,h)anthracene arithmetic means also exceeded their respective screening RBCs. Benzo(a)pyrene was detected in seven out of 25 samples at concentrations ranging from 37 ug/kg to 2,400 ug/kg. Dibenz(a,h)anthracene was detected only in one sample out of 25 (4 percent detection frequency) at a concentration of 290 ug/kg

While the maximum concentration of Total 2,3,7,8-TCDD Toxicity Equivalents exceeded the RBC, the arithmetic mean, which represents a more typical value for potential exposure, was less than the respective RBC. Total 2,3,7,8-TCDD Toxicity Equivalents were present in all 18 surface soil samples with concentrations ranging from 0.00044 ug/kg to 0.01605 ug/kg.

The maximum detects of Aroclor-1260, benzo(a)anthracene, benzo(b)fluoranthene, and indeno(1,2,3-cd)pyrene exceeded their respective screening RBCs. In contrast, the arithmetic means for these COPCs were each less than their respective screening RBCs, indicating that average exposure concentrations would be below screening criteria. Aroclor-1260 was detected in only one sample out of 26 (4 percent detection frequency) at a concentration of 630 ug/kg. Benzo(a)anthracene was detected in seven out of 25 samples at concentrations ranging from 40 ug/kg to 2500 ug/kg. Benzo(b)fluoranthene was detected in eight out of 25 samples at concentrations ranging from 49 ug/kg to 5,300 ug/kg. Indeno(1,2,3-cd)pyrene was detected in five out of 25 samples at concentrations ranging from 47 ug/kg to 1,100 ug/kg.

7.2.1.2 Soil* COPC Selection

The following substances were selected as COPCs in soil* (mixture of surface and subsurface soil) and are shown on Table 7.2:

- Total 2,3,7,8-TCDD Toxicity Equivalents
- Aroclor-1260
- Benz(a)anthracene
- Benzo(a)pyrene
- Benzo(b)fluoranthene
- Dibenz(a,h)anthracene
- Indeno(1,2,3-cd)pyrene

While the above 5 PAHs exhibited maximum concentrations above screening criteria, 13 additional PAHs were detected at levels below RBCs and therefore are not expected to contribute to significant site-related risks. 23 VOCs were detected at concentrations less than RBCs and at low frequencies of detection (from one to 12 out of 74 samples). Maximum concentrations of the metals aluminum, arsenic, iron, and vanadium exceeded their respective RBCs but all of these metals were found to be not above background in surface soil and subsurface soil based upon statistical tests shown in Tables 5.5 and 5.6.

While COPC selection was contingent upon maximum detected concentrations exceeding RBCs, from a risk-based perspective it is significant to note other descriptive statistics to better understand the range of detected concentrations that may affect risk for specific COPCs. For example, benzo(a)pyrene and

dibenz(a,h)anthracene arithmetic means also exceeded their respective screening RBCs. Benzo(a)pyrene concentrations ranged from 37 ug/kg to 140,000 ug/kg and dibenz(a,h)anthracene from 290 ug/kg to 13,000 ug/kg. Dibenz(a,h)anthracene was only detected in one percent (one out of 71) samples.

While the maximum concentration of Total 2,3,7,8-TCDD Toxicity Equivalents exceeded the respective RBC, note that the arithmetic mean, which represents a more typical value for potential exposure, was less than the respective RBC. Total 2,3,7,8-TCDD Toxicity Equivalents were present in all 20 soil* samples with concentrations ranging from 0.00044 ug/kg to 0.01605 ug/kg.

The maximum detected concentrations of Aroclor-1260, benzo(a)anthracene, benzo(b)fluoranthene, and indeno(1,2,3-cd)pyrene exceeded their respective screening RBCs. In contrast, the arithmetic means for these COPCs were each less than their respective screening RBCs, indicating that average exposure concentrations would be below screening criteria. Aroclor-1260 was detected in one sample out of 74 (detection frequency one percent) at a concentration of 630 ug/kg. The majority of the detected PAHs were found in 8 percent to 15 percent of samples and most spanned a broad range of concentrations - maximum concentrations were typically at least 25,000 ug/kg and minimum concentrations were generally below 100 ug/kg. Benzo(b)fluoranthene concentrations ranged from 46 to 300,000 ug/kg, benzo(a)anthracene from 40 ug/kg to 230,000 ug/kg, and indeno(1,2,3-cd)pyrene from 47 ug/kg to 36,000 ug/kg.

7.2.1.3 Groundwater COPC Selection

The following substances were selected as COPCs in groundwater and are shown on Table 7.3:

- Arsenic
- Iron
- Lead
- Manganese
- 2-Methylnaphthalene
- 4-Methylphenol
- Bis(2-ethylhexyl) Phthalate
- Naphthalene
- Pentachlorophenol
- 1,1,1-Trichloroethane
- 1,2,4-Trimethylbenzene
- 1,2-Dichloroethene (cis)

- 1,3,5-Trimethylbenzene
- Benzene
- Chloroform
- Tetrachloroethene
- Toluene
- Vinyl Chloride
- Xylene (Total)

The 10 VOCs listed above exhibited maximum concentrations above screening criteria. Ten additional VOCs were detected at levels below RBCs and therefore are not expected to contribute to significant site-related risks. Individual xylene isomers exceeded RBCs as well as total xylenes. Note that screening criteria and toxicity factors are identical for all xylene isomers so risk characterization can be performed without loss of specificity using only the sum of isomers, xylene (total), rather than individual isomers. This approach is more representative because it accounts for the cumulative exposure at each sample location.

Four metals exceeded RBCs and nine metals were below RBCs. Background tests were considered not applicable for groundwater due to a limited number of samples so COPC selection was based only on RBC comparison. However, site-related influence on metals concentrations is suggested based on information presented in the Nature and Extent of Contamination (Section 5) because the location of maximum metals concentrations is similar to locations where the organic plume concentrations are greatest. Arsenic and manganese were also detected in upgradient samples, but their maximum background concentrations were an order of magnitude less than maximum levels observed in onsite or downgradient wells. Lead was not detected in upgradient samples. The maximum upgradient iron concentration was one-fourth the maximum concentration found in onsite or downgradient wells.

While COPC selection was contingent upon maximum detected concentrations exceeding RBCs, from a risk-based perspective it is significant to note other descriptive statistics to better understand the range of detected concentrations that may affect risk for specific COPCs. For several chemicals, the arithmetic mean of groundwater concentrations exceeded the respective screening RBC; this was the case for arsenic, iron, manganese, 4-methylphenol, naphthalene, pentachlorophenol, 1,2,4-trimethylbenzene, 1,2-dichloroethene (cis), 1,3,5-trimethylbenzene, benzene, chloroform, tetrachloroethene, and vinyl chloride. Arsenic was detected in three out of 20 samples at concentrations ranging from 5.1 µg/L to 25.4 µg/L and iron was found in all 11 groundwater samples at concentrations ranging from 186 to 44,200 µg/L. Manganese was detected in all 18 samples at concentrations ranging from 9 to 4,990 µg/L. 4-Methylphenol was detected in two out of 19 samples at concentrations ranging from 88 to 300 µg/L. Naphthalene was detected in four out of 20 samples at concentrations ranging from 21 to 66 µg/L. Pentachlorophenol, which

is chemically unrelated to other classes of contaminants found at the FFTA Site, was detected in only one sample out of 19 (detection frequency 5 percent) at a trace concentration (2 µg/L). Several chlorinated VOCs and substituted benzenes were found in 10 percent to 33 percent of groundwater samples at low µg/L concentrations: 1,1,1-trichloroethane concentrations ranged from 2 to 340 µg/L, 1,2,4-trimethylbenzene from 5 to 65 µg/L, 1,2-dichloroethene (cis) from 1 to 460 µg/L, 1,3,5-Trimethylbenzene from 2 to 18 µg/L, benzene from 1 to 28 µg/L, chloroform from 1 to 7 µg/L, tetrachloroethene at 1 µg/L, vinyl chloride from 2 to 6 µg/L, and xylene (total) from 2 to 66 µg/L.

The maximum concentrations of lead, 2-methylnaphthalene, bis(2-ethylhexyl)phthalate, 1,1,1-trichloroethane, toluene, and xylene (total) exceeded their respective screening RBCs. Note that the arithmetic means were less than their respective screening RBCs, indicating that average exposure concentrations would be below screening criteria. 2-Methylnaphthalene was detected in four out of 20 samples at a concentration range of 5 to 35 µg/L. Bis(2-ethylhexyl)phthalate was detected in six out of 17 samples at a concentration range of 1 to 6 µg/L. 1,1,1-Trichloroethane was detected in 10 out of 21 samples at a concentration range of 2 to 340 µg/L. Toluene was detected in two out of 21 samples at concentrations of 12 and 83 µg/L. Xylene (total) was detected in 7 out of 21 samples at concentrations of 2 to 66 µg/L.

7.3 EXPOSURE POINT CONCENTRATIONS

In this risk assessment, an exposure point concentration (EPC) represents an estimated chemical concentration to which a receptor is assumed to be continuously exposed while in contact with an environmental medium. Using all the analytical results for related samples, an EPC was calculated for each COPC identified in each medium of concern at the FFTA Site. The EPC was calculated using the latest risk assessment guidance from EPA (1989a, 1991a, 1991b, 2000, 2002a, 2002b), Gilbert (1987), and Singh (1999). The shape of the data distribution and magnitude of the standard deviation for each COPC define the basis for calculating the upper 95 percent confidence limit on the mean concentration, which is the input concentration of a chemical used to estimate the site-associated risks.

Before EPCs are calculated, the analytical database is manipulated to remove unusable data points, average field duplicate results, and calculate statistics that determine the potential distributional shape of the data as previously discussed in Section 5.3.2, Data Pretreatment. The results of calculations of the shape of the data distribution are presented in distributional analysis tables in Appendix G. This information, together with the calculated magnitude of the standard deviation or standard deviation of log-transformed data for each COPC, defines the basis for calculating the upper 95 percent confidence limit on the mean concentration, which is the input concentration of a chemical used to estimate the site-associated risks.

7.3.1 Reasonable Maximum and Central Tendency Exposure EPCs

Two types of EPCs are possible for use in this risk assessment, reasonable maximum exposure (RME) EPCs and central tendency exposure (CTE) EPCs. RME is the exposure that is expected to represent a high end, but not worst-case, exposure in a given medium of concern. CTE is the exposure that is expected to represent an average exposure to receptors in a given medium of concern. The CTE EPC was selected to be the same as the RME EPC, in accordance with EPA Region III recommendations. Therefore, in this risk assessment, CTE analysis involves changes only to the input parameters for each exposure pathway and not the EPC. (Note: CTE analysis at FFTA was performed only for those exposure pathways where the estimated cancer risks are above 1×10^{-4} and the non cancer HIs based on the same target organ are above 1.0.)

7.3.2 EPC Calculations

EPCs are selected following a decision scheme that chooses from several options, depending upon how many data points exist, whether the data are normal or lognormally distributed, and the particular value for the standard deviation of logarithmically-transformed chemical concentrations. EPCs can be either the maximum value, 95 percent upper confidence limit on the mean of normally distributed data (95 percent UCL-N), the 95 percent H-statistic upper confidence limit on log transformed data (95 percent UCL-H), the Chebychev 95 percent confidence limit on logarithmically-transformed data using minimum variance unbiased estimates (95 percent UCL-Cheby-mv), the Chebychev 99 percent confidence limit on log transformed data using minimum variance unbiased estimates (99 percent UCL-Cheby-mv), the Chebychev 95 percent confidence limit assuming a nonparametric distribution (95 percent UCL-Cheby), the Chebychev 97.5 percent confidence limit assuming a nonparametric distribution (97.5 percent UCL-Cheby), or the Chebychev 99 percent confidence limit assuming a nonparametric distribution (99 percent UCL-Cheby). All the foregoing represent conservative estimates of the mean, each of which is appropriate under different circumstances.

EPCs for lead were calculated differently because the model is designed to accept the mean lead value and estimate the upper percentile of blood-lead concentrations from this quantity. Therefore, in the case of lead, the candidate RME EPC was selected as the arithmetic mean of the lead concentration (for a normal distribution) in every case.

The calculation of an EPC involves two steps. First, the distribution of the data was evaluated as described in the preceding section. Then, based on the distribution of the data, an EPC was either calculated or selected.

Several important assumptions were used to evaluate the distribution of the data (Section 5.3.2):

- The distribution of a data set was classified as lognormal, normal, or unknown (i.e., nonparametric) using the results of the Shapiro Wilk W-test. If the W test accepted the hypothesis of a normal distribution, then the W test using the logarithmic conversion was not applicable and the estimation of the upper 95th percentile confidence limit on the mean was based upon a normal distribution and standard deviation. If the normal distribution did not pass the W-test, then the W-test was evaluated for a logarithmic transformation of data. If the data were neither normal nor lognormally distributed, but the standard deviation of the logarithms was less than 0.5, a normal distribution was assumed. Otherwise, if neither distribution matched the data set of interest, a nonparametric distribution was assumed.
- If fewer than three samples were available in the area or medium of concern, estimation of the distributional shape was not possible and the 95 percent UCL was not estimated.
- If fewer than five samples were available, the determination of distributional shape and the estimation of statistical upper confidence limits were not considered reliable or accurate because of inadequate sample size, and the maximum value was selected as the EPC in this case.

If the data were determined to be normally distributed, then the standard deviation of the sample set and the student's t-value were used to calculate the one-sided 95 percent UCL, as follows:

First, the standard deviation of the sample set was determined:

$$S = \sqrt{\sum \left(\frac{(X_i - \mu)^2}{n - 1} \right)}$$

where:

S	=	Standard deviation of the data
X_i	=	Individual sample value
μ	=	Arithmetic mean of the n samples
n	=	Number of samples

The one-sided upper 95 percent confidence limit (95 percent UCL-N) was calculated as follows:

$$95\%UCL - N = \mu + \frac{(t \times S)}{\sqrt{n}}$$

where:

- S = Standard deviation of the data
- t = One-sided t distribution factor
- μ = Arithmetic mean of the n samples
- n = Number of samples

If the data were considered to be lognormally distributed, then the EPC was selected following a decision scheme listed below. The EPC was based on either the maximum value, the 95 percent H-statistic upper confidence limit on log transformed data (95 percent UCL-H), the Chebychev 95 percent confidence limit on log transformed data using minimum variance unbiased estimates (95 percent UCL-Cheby-mv), the Chebychev 99 percent confidence limit on log transformed data using minimum variance unbiased estimates (99 percent UCL-Cheby-mv), the Chebychev 95 percent confidence limit (95 percent UCL-Cheby), or the Chebychev 99 percent confidence limit (99 percent UCL-Cheby). The following table presents the type of upper confidence limit chosen according to the number of data points and the observed value for the log standard deviation:

Statistical UCL Methods for Lognormally Distributed Data

No. of Data Points	Log Standard Deviation	Type of UCL
$N < 5$	--	Maximum value
$N \geq 5$	$0.5 \leq S < 1.0$	95% UCL-H
$5 \leq N < 25$	$1.0 \leq S < 1.5$	95% UCL-Cheby-mv
$N \geq 25$	$1.0 \leq S < 1.5$	95% UCL-H
$5 \leq N < 20$	$1.5 \leq S < 2.0$	99% UCL-Cheby-mv
$20 \leq N < 50$	$1.5 \leq S < 2.0$	95% UCL-Cheby-mv
$N \geq 50$	$1.5 \leq S < 2.0$	95% UCL-H
$5 \leq N < 25$	$2.0 \leq S < 2.5$	99% UCL-Cheby-mv
$25 \leq N < 70$	$2.0 \leq S < 2.5$	95% UCL-Cheby-mv
$N \geq 70$	$2.0 \leq S < 2.5$	95% UCL-H
$5 \leq N < 30$	$S \geq 2.5$	Maximum of quantities: 99% UCL-Cheby-mv or 99% UCL-Cheby
$30 \leq N < 70$	$S \geq 2.5$	Maximum of quantities: 95% UCL-Cheby-mv or 95% UCL-Cheby
$N \geq 70$	$S \geq 2.5$	95% UCL-H

For data considered to be lognormal, the standard deviation of the log-transformed sample set was determined as follows:

$$S = \sqrt{\sum \left(\frac{(X_i - \mu)^2}{n-1} \right)}$$

where:

S	=	Standard deviation of the log-transformed data
X_i	=	Individual sample value (log-transformed)
μ	=	Arithmetic mean of the log-transformed n samples
n	=	Number of samples

If the EPC was based on the 95 percent H-statistic upper confidence limit on log transformed data (95 percent UCL-H), this quantity was then calculated as follows:

$$95\%UCL - H = e^{\left[\mu + 0.5S^2 + \left(\frac{SH}{\sqrt{n-1}} \right) \right]}$$

where:

e	=	constant (base of the natural log, equal to 2.718)
μ	=	Arithmetic mean of the log-transformed data
H	=	H-statistic (e.g., from table published in Gilbert, 1987)
S	=	Standard deviation of the log-transformed data
n	=	Number of samples

Most of the lognormal cases listed above required use of upper confidence limits based on Chebychev equations using minimum variance unbiased estimates (95 percent UCL-Cheby-mv or 99 percent UCL-Cheby-mv). In these cases, the mean and standard deviation were calculated in a different manner. For the Chebychev confidence limits, the following equations were used to calculate the minimum variance unbiased estimates of the population's mean (mean-T) and standard deviation (sigma-mv) for lognormally distributed data, according to Gilbert (1987):

$$\text{Mean-T} = \exp\left\{ \bar{L} + Q_n(s_y^2/2) \right\}$$

Where: \bar{L} = arithmetic mean of log-transformed data
 s_y = standard deviation of log-transformed data
 $Q_n(t)$, with $t = s_y^2/2$, is the infinite series:

$$Q_n(t) = 1 + (n-1)t/n + (n-1)^3 t^2 / (2!n^2(n+1)) + (n-1)^5 t^3 / (3!n^3(n+1)(n+3)) + (n-1)^7 t^4 / (4!n^4(n+1)(n+3)(n+5)) + \dots$$

$$\text{Sigma-mv} = \exp\{2 \cdot \bar{L} \cdot [Q_n(2s_y^2) - Q_n(s_y^2(n-2)/(n-1))]\}$$

Where: \bar{L} = arithmetic mean of log-transformed data

s_y = standard deviation of log-transformed data

$Q_n(t)$, with $t = 2s_y^2$ or $t = s_y^2(n-2)/(n-1)$, is the infinite series:

$$Q_n(t) = 1 + (n-1)t/n + (n-1)^3 t^2 / (2!n^2(n+1)) + (n-1)^5 t^3 / (3!n^3(n+1)(n+3)) + (n-1)^7 t^4 / (4!n^4(n+1)(n+3)(n+5)) + \dots$$

The Chebychev upper confidence limits were then estimated using the following equations, according to Singh (1999):

$$95\%UCL - Chebymv = meanT + sigmamv \times \sqrt{\frac{1}{0.05 \times N}}$$

$$99\%UCL - Chebymv = meanT + sigmamv \times \sqrt{\frac{1}{0.01 \times N}}$$

If the data were considered to be neither normally nor lognormally distributed, then the EPC was selected following a decision scheme listed below. If the standard deviation of the log transformed data was less than 0.5, then the EPC was calculated assuming a normal distribution and the 95 % UCL using the t-distribution, as delineated earlier. Otherwise, the EPC was based on either the Chebychev 95 percent confidence limit (95 percent UCL-Cheby), the Chebychev 97.5 percent confidence limit (97.5 percent UCL-Cheby), or the Chebychev 99 percent confidence limit (99 percent UCL-Cheby). The following table presents the type of upper confidence limit chosen according to the number of data points and the observed value for the log standard deviation:

Statistical UCL Methods for Nonparametrically Distributed Data

No. of Data Points	Log Standard Deviation	Type of UCL
$N < 5$	--	Maximum value
$N \geq 5$	$S < 0.5$	95% UCL-N
$N \geq 5$	$0.5 \leq S < 1.0$	95% UCL-Cheby
$N \geq 5$	$1.0 \leq S < 2.0$	97.5% UCL-Cheby
$N \geq 5$	$S \geq 2.0$	99% UCL-Cheby

For data considered to be nonparametric, the standard deviation of the log-transformed sample set was determined as follows:

$$S = \sqrt{\sum \left(\frac{(X_i - \mu)^2}{n-1} \right)}$$

where:

S	=	Standard deviation of the log-transformed data
X _i	=	Individual sample value (log-transformed)
μ	=	Arithmetic mean of the log-transformed n samples
n	=	Number of samples

Wherever the nonparametric case required use of upper confidence limits based on Chebychev equations, the EPC was based on either the Chebychev 95 percent, 97.5 percent, or 99 percent confidence limits. In these cases, the mean and standard deviation were calculated in the conventional manner used for a normal distribution.

The Chebychev upper confidence limits were then estimated using the following equations, according to Singh (1999):

$$95\%UCL - Cheby = \mu + S \times \sqrt{\frac{1}{0.05 \times N}}$$

$$97.5\%UCL - Cheby = \mu + S \times \sqrt{\frac{1}{0.025 \times N}}$$

$$99\%UCL - Cheby = \mu + S \times \sqrt{\frac{1}{0.01 \times N}}$$

where:

S	=	Standard deviation of the original (not transformed) data
μ	=	Arithmetic mean of the original (not transformed) sample data
N	=	Number of samples

7.3.3 Optional Use of Maximum Observed Concentration

The preceding decision scheme was used to generate statistical estimates of the UCL for each COPC in accordance with recent guidance (EPA, 2002a, 2002b). In most, but not all cases, this approach produces statistical UCL estimates that are not unrealistically high, which is one of the main goals of the recent guidance. In contrast, earlier guidance (EPA, 1992e) specified that the H-distribution be applied to UCLs for all lognormal and nonparametric cases and had a drawback of frequently generating extremely high UCLs greater than the maximum detected concentration.

To prevent unrealistically high EPCs at the FFTA site, if the statistical UCL was greater the maximum observed concentration, then the maximum was selected as the default EPC. This optional step is allowed in current guidance (EPA, 2002a), but there is a caveat that this may yield a reasonable EPC estimate only if the sample size is sufficiently large and samples have been collected at random from the exposure unit.

7.3.4 EPCs Selected for FFTA Groundwater, Surface Soil, and Soil*

Tables 7.4, 7.5, and 7.6 present the RME and CTE EPCs selected for FFTA surface soil, soil* (mixture of surface and subsurface soil), and groundwater, respectively. The columns labeled rationale for selecting the EPC indicate the applicable factors that determine the EPC decision for each COPC, including number of data points, log standard deviation, and comparison of distributional test scores for normal data (Wno), lognormal data (Wlg), and lookup table values (Wt). Appendix G contains comparisons of various UCL estimates for each detected chemical and associated values of the aforementioned parameters involved in the UCL decision scheme.

7.4 EXPOSURE ASSESSMENT

The exposure assessment evaluates the potential for human exposure to the chemicals detected in the environmental media of concern at the FFTA Site investigated during the RI. This section presents a characterization of the exposure setting, characterizes the exposed populations, identifies actual or potential exposure routes, and summarizes the methods used to generate exposure estimates. To determine whether there is an actual or potential exposure, the most likely pathways of contaminant release and transport, as well as the human and environmental activity patterns, are considered. A complete exposure pathway has three components: a source, a route of transport, and an exposure point for receptors. These components are addressed in this section.

7.4.1 Characterization of the Exposure Setting

A general description of the facility, its setting, and its surroundings is provided in Section 2.0 of the work plan (TtNUS, 2003a). Section 3.0 of the work plan includes a characterization of FFTA site conditions, including physiography and topography, climate, soil, surface-water hydrology, and public water supply and use.

7.4.2 Potential Receptors

The receptors chosen for the FFTA site are presented in this section. All of the receptors listed below are applicable to the FFTA site because all selected media were sampled at the site. The potential exposure scenarios are as follows:

- **Current and Future Industrial Worker** - The full-time onsite worker is an adult who works at WFF year round. This receptor is potentially exposed via incidental ingestion of, dermal contact with, and particulate dust inhalation of COPCs in surface soil. Examples of potential activities include groundskeeping and lawn maintenance, installation or maintenance of airfield equipment, and utility or roadwork. Inhalation of VOCs from vapor emissions from soil was not considered a significant pathway of exposure because VOCs were detected infrequently and at low concentrations in soil at the FFTA. Groundwater ingestion was not evaluated because the FFTA is located adjacent to an active runway where no buildings are present or expected to be constructed. In addition, it is unlikely or not reasonable to assume that potable water wells for commercial/industrial purposes would be installed at the FFTA because the shallow aquifer is a low-yielding aquifer and use of the aquifer is controlled and restricted by Virginia through issuances of permits under the Groundwater Management Act. In addition public water is available at the site. The unlikely use of groundwater is evaluated under the residential scenario.
- **Future Construction Worker** - A future construction worker is an adult who is assumed to work at WFF involved with any type of excavation activity. This receptor is potentially exposed to disturbed soil represented by a mixture of surface soil and subsurface soil. The future construction worker may be exposed via incidental ingestion of, dermal contact with, and particulate dust inhalation of COPCs in disturbed soil. Additionally, this receptor is potentially exposed via dermal contact with contaminants in groundwater and inhalation of VOCs in shallow groundwater during excavation activities. Inhalation of VOCs from vapor emissions from soil was not considered a significant pathway of exposure because VOCs were detected infrequently and at low concentrations in soil at the FFTA.

- Future Resident - A future resident is a person who will live in a residence located at or near WFF in a hypothetical future scenario. This receptor occupies a residence as a child (from age 0 - 6 years) and as an adult (for 24 years exposure duration). This receptor is potentially exposed to COPCs in groundwater via tap water ingestion, dermal contact while bathing, and inhalation of VOCs present in vapors generated during showering (adult resident only). In addition, the future resident is potentially exposed via incidental ingestion of, dermal contact with, and particulate dust inhalation of COPCs in surface soil. Inhalation of VOCs from vapor emissions from soil is not considered a significant pathway of exposure because VOCs were detected infrequently and at low concentrations in soil at the FFTA. Non-cancer risks were estimated separately for child versus adult, whereas, cancer risks were considered cumulative (risks were summed over child and adult periods of exposure). Additionally, potential exposure to disturbed soil that is a mixture of surface and subsurface soils as a result of construction or landscaping activities was addressed.
- In accordance with the FFTA Supplemental RI work plan, potential exposures to the surface water pathway near the FFTA were not considered in this risk assessment. The only exposures to this pathway near the FFTA would involve contact with surface water or sediment within an unnamed tributary or Little Mosquito Creek. Risk assessments of both areas were previously presented within the WFF Site 14 and 15 RI report that has been submitted to the EPA and VADEQ and is undergoing review, comment, and finalization. WFF Sites 14 and 15 have a greater potential for direct impact to these area surface waters than the FFTA and for this reason, no further sampling or assessment was proposed as part of the FFTA RI.

7.4.3 Exposure Estimates

The estimation routes, methods, and models presented in this section are consistent with current EPA risk assessment guidance (EPA, 1989a, 1991b, 1992c, 1993a, 1994b, 1995, 1996b, 1996c, 1997b, and 2001a). Exposure estimates associated with each exposure route are presented below. All exposure scenarios incorporate RME and CTE EPCs in the estimation of intakes. There are three environmental media for WFF through which potential receptors (see previous section) can be either directly or indirectly exposed to site-related COPCs: groundwater, surface soil, and soil* (disturbed soil with a composition represented by a mixture of surface soil and subsurface soil). Note that disturbed soil was considered to represent only samples collected at depths less than 15 feet, consistent with VADEQ exposure assessment guidelines that are based upon a 15 foot depth of soil excavation for building foundations.

Noncarcinogenic risks are estimated using the concept of an average annual exposure. The intake incorporates terms describing the exposure time and/or frequency that represent the number of hours per day and the number of days per year that exposure occurs. This is used along with the "averaging time",

which converts the total annual exposure to an average daily dose by dividing by 365 days per year of exposure. Noncarcinogenic risks for some exposure routes (e.g., soil) are generally greater for children than for adults because of differences in body weight and intake. Carcinogenic risks, on the other hand, are estimated as an incremental lifetime risk and, therefore, incorporate terms to average the exposure duration (years) over the course of a lifetime (70 years).

7.4.4 Groundwater Exposure Estimates

Four potential exposure routes are associated with groundwater at the FFTA:

- Ingestion
- Dermal absorption
- Inhalation of airborne vapors during showering
- Inhalation of VOCs in shallow groundwater during excavation

For groundwater, future construction workers were evaluated only for dermal absorption and inhalation of VOCs in shallow groundwater during excavation because incidental ingestion of groundwater during construction is not expected to result in significant intake of groundwater. Several exposure pathways were evaluated for the future resident: Tap water ingestion and dermal absorption while bathing or showering were evaluated for a future residential child, future residential adult, and future lifetime resident (cancer risk only). Additionally, for groundwater, inhalation of airborne vapors during showering was evaluated for a future residential adult. While future residential land use is not anticipated at the FFTA, the degree of exposure to contaminants by residential receptors would be greater than with other exposure scenarios, so that a residential exposure evaluation conservatively assesses whether there is any need for institutional controls or other remedial measures under all possible future exposure scenarios that might occur at the FFTA.

Tables 7.7 and 7.8 present the RME and CTE input parameters selected for groundwater ingestion and dermal exposure pathways, respectively, for a residential child. Tables 7.9 and 7.10 present the RME and CTE input parameters selected for groundwater ingestion and dermal exposure pathways, respectively, for a residential adult. Table 7.11 presents the RME and CTE input parameters selected for the groundwater inhalation exposure pathway for a residential adult. Table 7.12 presents the RME and CTE input parameters selected for dermal exposure to shallow groundwater during excavation by a construction worker. Table 7.13 presents the RME and CTE input parameters selected for the inhalation of VOCs in shallow groundwater during excavation.

For the groundwater exposure pathways, the following items are noteworthy:

- Chemical-specific permeability constants associated with modeling of dermal absorption are not shown on the exposure input tables. The dermal permeability constants were obtained from EPA RAGS Part E, Interim Dermal Exposure Guidance (EPA, 2001a) and are presented in Appendix G for each COPC.
- Derivations of the surface areas used in all dermal exposure equations in this risk evaluation for each potential receptor are based on surface areas for available body parts. Surface areas were extracted from dermal guidance (EPA, 2001a) for each potential receptor:
- Chemical-specific parameters associated with modeling of inhalation of airborne vapors during showering are not shown on the exposure input tables. Henry's Law constants and other parameters used in this model are obtained from several sources, including EPA (1996c) and Foster and Chrostowski (1987) and are presented in Appendix G for each COPC. Henry's Law constants published for 25 degrees C were corrected to account for an elevated showering temperature of 45 degrees C (EPA, 2001b and Sander, 1999).
- The ASTM Guidance (Designation E 1739 – 95) presents the model parameters for determination of a volatilization rate of VOCs from groundwater into air. The following equations were utilized:

$$VF = \frac{H'}{1 + \left[\frac{U_{air} * S_{air} * L_{GW}}{W * D_{EFF-WS}} \right]} * 10^3 \text{ L/m}^3$$

Where:

- VF = Volatilization Factor for VOC in groundwater
- H' = dimensionless Henry's Law constant for VOC at 25 degrees C
- U_{air} = 470 cm/sec = wind speed above ground surface in the mixing zone
- S_{air} = 200 cm = ambient mixing zone
- L_{GW} = 0.1 cm = depth to groundwater, h_v+h_{cap}
- h_v = 0 cm = thickness of the vadose zone
- h_{cap} = 0.1 cm = thickness of the capillary fringe
- D_{EFF-WS} = effective diffusion between GW and Soil for VOC
- W = 1500 cm = width of the source parallel to groundwater flow direction

$$D_{EFF-WS} = (h_{cap} + h_v) * \left[\frac{h_{cap}}{D_{EFF-cap}} + \frac{h_v}{D_{EFF-s}} \right]^{-1}$$

Where:

D_{EFF-WS} = effective diffusion between GW and Soil for VOC

h_v = 0 cm = thickness of the vadose zone

h_{cap} = 0.1 cm = thickness of the capillary fringe

$D_{EFF-cap}$ = effective diffusion through capillary fringe for VOC

D_{EFF-s} = cm²/sec = effective diffusion in soil based on vapor-phase concentration, chemical specific

Because h_v is equal to zero, this equation reduces to show that the effective diffusion between groundwater and surface soil (D_{EFF-WS}) is equal to the effective diffusion through the capillary fringe ($D_{EFF-cap}$).

$$D_{EFF-cap} = \left[D^{air} * \frac{Oa_{cap}^{3.33}}{Ot^2} \right] + \left[D^{wat} * \frac{1}{H'} * \left(\frac{Ow_{cap}^{3.33}}{Ot^2} \right) \right]$$

Where:

$D_{EFF-cap}$ = effective diffusion through capillary fringe for VOC

D^{air} = cm²/sec = diffusion coefficient in air for VOC

D^{wat} = cm²/sec = diffusion coefficient in water for VOC

H' = dimensionless Henry's Law constant for VOC

Oa_{cap} = 0.038 cm³-air/cm³-soil = volumetric air content in capillary fringe soils (equals O_t minus Oa_{cap})

Ow_{cap} = 0.342 cm³-H₂O/cm³-soil = volumetric water content in capillary fringe soils

O_t = 0.38 cm³/cm³-soil = total soil porosity

7.4.5 Surface Soil and Disturbed Soil Exposure Estimates

Three potential exposure routes are associated with direct exposure to surface soil and disturbed soil at the FFTA, including:

- Ingestion
- Dermal absorption
- Inhalation of fugitive dust

For disturbed soil (with a composition represented by a mixture of surface soil and subsurface soil), incidental ingestion, dermal absorption, and inhalation of fugitive dust were evaluated for a future

residential child, future residential adult, future lifetime resident (cancer risk only), and a future adult construction worker. For surface soil, ingestion, dermal absorption, and inhalation of fugitive dust were evaluated for a future residential child, future residential adult, future lifetime resident (cancer risk only), and a current/future adult industrial worker. For soil, VOCs were detected infrequently at trace levels. Therefore, inhalation exposure to VOC emissions from soil is expected to be negligible at the FFTA, with ingestion and dermal absorption contributing to the majority of potential risks.

Tables 7.14, 7.15, and 7.16 present the RME and CTE input parameters for a residential child exposed to COPCs in soil (either surface soil or disturbed soil) via incidental ingestion, dermal contact, and inhalation of particulate dusts, respectively. Tables 7.17, 7.18, and 7.19 present the RME and CTE input parameters for a residential adult exposed to COPCs in soil (either surface soil or disturbed soil) via incidental ingestion, dermal contact, and inhalation of particulate dusts, respectively. Tables 7.20, 7.21, and 7.22 present the RME and CTE input parameters for an industrial worker (groundskeeper or maintenance worker) exposed to COPCs in surface soil via incidental ingestion, dermal contact, and inhalation of particulate dusts, respectively. Tables 7.23, 7.24, and 7.25 present the RME and CTE input parameters for a construction worker exposed to COPCs in disturbed soil via incidental ingestion, dermal contact, and inhalation of particulate dusts, respectively. For surface soil and disturbed soil exposure pathways, the following items are noteworthy:

- Chemical-specific absorption factors associated with dermal absorption were not shown on the exposure input tables. These values are provided in EPA dermal exposure guidance (EPA, 2001a) and specific EPA Region III guidance (EPA, 1995) and are presented in Section 7.5.
- The site-specific particulate emission factor (PEF) associated with modeling inhalation of fugitive dust was derived based on information presented in EPA's Soil Screening Guidance (EPA, 1996b) and site-specific information. The PEF was calculated as follows:

$$PEF(m^3 / kg) = \frac{Q/C \times 3600s / h}{E_{10}}$$

where:

PEF	=	3.68 x 10 ⁸ m ³ /kg	= Site-specific particulate emission factor for FFTA Site
Q/C	=	90.24 (g/m ² -s)/(kg/m ³)	= Inverse of the mean conc. at the center of a 0.5 acre source (Q/C climate zones table in EPA, 1996b)
E ₁₀	=	0.036 x (1-V) x (U _m /Ut) ³ x F(x)	= Particulates less than 10 microns (PM ₁₀) average annual emission flux (g/m ² -hr)
V	=	0.8	= Fraction of Vegetative cover
U _m	=	4.7 m/sec	= Mean Annual Windspeed at 7 m (Table 4-1, Cowherd)

$U_t = 8.52 \text{ m/s}$ = Threshold value of windspeed at 7 m for FFTA
(from equation, below)

$F(x) = 0.73$ = Function of x , where $x = 0.886 \times U_t/U_m = 1.61$
(From Figure 4-3, Cowherd)

$U_t = U^* \times (1/0.4) \times \ln(z/z_0)$

where:

$U_t = 8.52 \text{ m/s}$ = Wind speed at height z for FFTA
 $z = 700 \text{ cm}$ = Height above surface (Cowherd)
 $z_0 = 3 \text{ cm}$ = Roughness height for site conditions: FFTA assume 3 cm
for grassland (Figure 3-6, Cowherd)
 $U^* = 0.625 \text{ m/s}$ = Friction velocity for assumed particle size 0.5mm
(Figure 3-4, Cowherd), corrected $\times 1.25$ (EPA, 1996b)

- Derivations of the surface areas used in all dermal exposure equations for each potential receptor were based on surface areas for available body parts. Surface areas were compiled from EPA sources (EPA, 2001a and 1997b) and are presented on the aforementioned tables for each receptor.

7.4.6 Blood-Lead Modeling

As outlined in OSWER Directive 9355.4-12, EPA (1994a) has developed an approach to evaluating lead risks that recognizes the multimedia nature of lead exposures, incorporating absorption and pharmacokinetic information. Research has been conducted concerning lead intake and resultant blood-lead levels. Determinations of lead uptake from soil and drinking water were considered. Potential blood-lead level increases are presented in Section 7.6, along with the potential implications of blood-lead results for residential children and industrial adults (pregnant females). The following discussion presents information that is useful in estimating lead exposure.

No threshold has been defined for effects related to blood-lead increases. Effects below blood-lead levels of 10 $\mu\text{g/dL}$ are difficult to define. Inhibition of certain enzymes involved in red blood cell metabolism has been reported to occur at 10 to 15 $\mu\text{g/dL}$ and possibly lower. Small increases in blood pressure have been observed in adults with blood-lead levels down to 7 $\mu\text{g/dL}$ (EPA, 1994b). The most sensitive subpopulation to effects below 7 $\mu\text{g/dL}$ would be infants, whose early neurological development can be affected by blood-lead concentrations reportedly down to 5 $\mu\text{g/dL}$ (EPA, 1994b). Lead is also a

fairly common environmental contaminant and, for this reason, typical blood-lead levels in the population at large may already exceed the concentrations discussed here.

For drinking water exposure, children 0 through 6 months old are expected to experience blood-lead increases at the rate of 0.26 ug/dL per ug/L lead in water up to 15 ug/L and at the rate of 0.04 ug/dL for every ug/L lead in water above 15 ug/L (EPA, 1994b). For older children, the ratio is 0.12 ug/dL blood lead per ug/L lead in water up to 15 ug/L and 0.06 ug/dL for every ug/L lead in water above 15 ug/L (EPA, 1994b). For adults, the ratio is approximately 0.06 ug/dL blood lead per ug/L in water (EPA, 1994b). Dietary intake of lead is assumed to produce increases of 0.02 to 0.04 ug/dL blood lead per ug/day ingested by adults and 0.16 ug/dL blood lead per ug/day ingested by infants (EPA, 1986a). Blood-lead levels are estimated to increase by 0.6 to 6.8 ug/dL per 1,000 mg/kg lead in soil (EPA, 1986a).

Blood-lead levels in residential children (age 0 to 6) were estimated using the Integrated Exposure and Uptake Biokinetic (IEUBK) Model (version 1.0) developed by EPA (EPA, 1994b). The model was applied using the average groundwater and either surface soil or disturbed soil concentrations in each applicable medium of concern where lead was selected as a COPC at the FFTA Site.

For the assessment of ingestion of lead in groundwater and either disturbed soil or surface soil by residential children, default values in the model were used to represent background lead concentrations in air, house dust, and the level of maternal contribution. Additionally, the model's default values were used to represent respiratory rate, soil and water ingestion rates, and the percent of lead absorption by the various exposure routes. The only site-specific factors put into the IEUBK Model were the average lead concentration in groundwater and soil.

The output of the IEUBK Model is a histogram that presents the estimated percentage of residential children (age 0 through 6 years) with a blood-lead level above 10 ug/dL (considered to be the threshold significance level above which adverse effects cannot be ruled out). When the percentage of the population estimated to have blood-levels above 10 ug/dL is greater than five percent, then EPA considers the potential for adverse effects to be significant (EPA, 1994a). These histograms, along with input information particular to each run of the IEUBK model, are presented in Appendix G, along with the estimated percentages of residential children (age 0 through 6 years) with blood-lead levels above 10 ug/dL. Uncertainties associated with the IEUBK model are discussed in Section 7.7.

Noncarcinogenic risks for industrial worker exposures to lead in surface soil or disturbed soil can be estimated using the Interim Approach to Assessing Risks Associated with Adult Exposures to Lead in Soil (EPA, 1996b). The model is based on a biokinetic slope factor that estimates fetal blood lead concentration in women exposed to lead in contaminated media. A simplified (linear) representation of

lead biokinetics is used to predict quasi-steady state blood lead concentrations among adults who have relatively steady patterns of lead exposure (exposure duration of at least 90 days and exposure frequencies greater than once per week). The intake assumptions used in the model are average lead concentrations in each in soil, the ingestion rate, and an exposure frequency. Note that lead was not selected as a COPC in surface soil or disturbed soil so no estimates were generated.

The biokinetic model input parameters are a biokinetic slope factor (ug/dl adult blood lead per ug/day uptake) of 0.4, a constant of proportionality between fetal blood lead concentration at birth and maternal blood lead concentration of 0.9, a soil absorption factor of 0.12 (equal to the product of the relative bioavailability of 0.6 and a soluble lead absorption factor of 0.2), a background blood lead (typical concentration for women of child bearing age not exposed to the site) of 1.7 ug/dl, and a geometric standard deviation ranging from 1.8 (representative of homogeneous population demographics). Blood lead predictions can provide an estimate of the 95th percentile of blood lead concentrations among fetuses born to women having site exposures.

7.5 TOXICITY ASSESSMENT

Toxicity Assessment identifies the potential health hazards associated with exposure to each of the COPCs. A toxicological evaluation characterizes the inherent toxicity of a compound. The literature indicates that the COPCs have the potential to cause carcinogenic and/or noncarcinogenic health effects in humans. Although the COPCs may cause adverse health effects, dose-response relationships and the potential for exposure must be evaluated before the risks to receptors can be determined. Dose-response relationships correlate the magnitude of the intake with the probability of toxic effects, as discussed below. Toxicity information for the COPCs in groundwater, surface soil, and disturbed soil at the FFTA Site are presented in Tables 7.26, 7.27, 7.28, and 7.29 (RAGs D Tables 5.1, 5.2, 6.1, and 6.2, respectively).

An important component of the risk assessment process is the relationship between the intake of a compound (the amount of a chemical that is absorbed by a receptor) 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 can be quantified. The published information of doses and responses is used in conjunction with information on the nature and magnitude of human exposure to develop an estimate of potential health risks.

Dose-response values [reference doses (RfDs) and slope factors (SFs)] have been developed by EPA and other sources for many organics and inorganics. This section provides a brief description of these parameters.

7.5.1 Reference Doses

The RfD is developed by EPA for chronic and/or subchronic human exposure to hazardous chemicals and is based solely on the noncarcinogenic effects of chemical substances. Subchronic RfDs are specifically developed to be protective for a portion of a lifetime exposure to a compound (as a Superfund program guideline, short term). Chronic RfDs are specifically developed to be protective for long-term exposure to a compound (as a Superfund program guideline, long term). The RfD is usually expressed as a dose (mg) per unit body weight (kg) per unit time (day). It is generally derived by dividing a No-Observed-(Adverse)-Effect-Level (NOAEL or NOEL) or a Lowest-Observed-Adverse-Effect-Level (LOAEL) by an appropriate uncertainty factor. NOAELs, etc. are determined from laboratory or epidemiological toxicity studies. The uncertainty factor is based on the availability of toxicity data.

Uncertainty factors are generally applied as multiples of 10 to represent specific areas of uncertainty in the available data. A factor of 10 is used to account for variations in the general population (to protect sensitive subpopulations), when test results from animals are extrapolated to humans (to account for interspecies variability), when a NOAEL derived from a subchronic study (instead of a chronic study) is used to develop the RfD, and when a LOAEL is used instead of a NOAEL. In addition, EPA reserves the use of a modifying factor of up to 10 for professional judgment of uncertainties in the database not already accounted for. The default value of the modifying factor is 1.

The RfD incorporates the surety of the evidence for chronic human health effects. Even if applicable human data exist, the RfD (as diminished by the uncertainty factor) still maintains a margin of safety so that chronic human health effects are not underestimated. Thus, the RfD is an acceptable guideline for evaluation of noncarcinogenic risk, although the associated uncertainties preclude its use for precise risk quantitation. Oral and dermal RfDs, primary target organs, uncertainty/modifying factors, and sources of noncancer toxicity information for COPCs are provided in Table 7.26 (RAGs D Table 5.1). Inhalation RfDs, primary target organs, uncertainty/modifying factors, and sources of toxicity information for selected COPCs in soil are provided in Table 7.27 (RAGs D Table 5.2). Inhalation RfDs (mg/kg/day) were derived from inhalation reference concentrations (RfCs) (mg/m³) by dividing by 70 kg (an assumed human body weight), multiplying by 20 m³/day (an assumed human inhalation rate), and adjusting by an appropriate absorption factor (EPA, 1997a).

Target organ data have been extracted from the Integrated Risk Information System (IRIS; EPA, 2003), Health Effect Assessment Summary Tables (HEAST; EPA, 1997a), or other applicable sources. Only the target organs that are affected in the applicable study in which the RfD was derived have been included in Tables 7.26 and 7.27 (RAGs D Tables 5.1 and 5.2, respectively).

Noncarcinogenic risks for lead were not quantified and compared to RfDs, because EPA has implemented an approach to evaluating lead risks that does not provide a single-point estimate output. Instead, potential lead exposures are evaluated using a biokinetic model to estimate expected blood-lead increases. The blood-lead model is discussed in Section 7.4.6. A discussion of the results of the blood-lead model estimates is presented in Section 7.6.

7.5.2 **Cancer Slope Factors (SFs)**

SFs are applicable for estimating the lifetime probability (assumed 70-year lifespan) of human receptors developing cancer as a result of exposure to known or potential carcinogens. This factor is generally reported in units of 1/(mg/kg/day) and is derived through an assumed low-dosage linear relationship of extrapolation from high to low dose responses determined from animal studies. The value used in reporting the slope factor is the upper 95 percent confidence limit.

Oral and dermal SFs, weight of evidence, and sources of toxicity information for selected COPCs in groundwater, surface soil, and disturbed soil are provided in Table 7.28 (RAGs D Table 6.1). Inhalation SFs, weight of evidence, and sources of toxicity information for selected COPCs in groundwater, surface soil, and disturbed soil are provided in Table 7.29 (RAGs D Table 6.2). Inhalation SFs (mg/kg/day)⁻¹ were derived from inhalation unit risks (ug/m³)⁻¹ by multiplying by 70 kg (an assumed human body weight), dividing by 20 m³/day (an assumed human inhalation rate), and multiplying by the appropriate conversion factor (1000 ug/mg) (EPA, 1997a).

Carcinogenic risks for lead were not quantified, because EPA has not published a SF for inorganic lead. Instead, potential lead exposures were evaluated using a biokinetic model to estimate expected blood-lead increases. A discussion of these results is presented in Section 7.6.

7.5.3 **EPA Weight of Evidence**

The weight-of-evidence designations indicate the preponderance of evidence regarding carcinogenic effects in humans and animals. The categories are defined as follows (EPA, 1992a):

WEIGHT OF EVIDENCE CATEGORY	DEFINITION
A	Known human carcinogen
B1	Probably human carcinogen, limited human data are available
B2	Probable human carcinogen, sufficient animal data are available but inadequate human data are available
C	Possible human carcinogen
D	Not classifiable as to human carcinogenicity
E	Evidence of noncarcinogenicity in humans

7.5.4 Adjustment of Dose-Response Parameters for Dermal Exposure

Risks associated with dermal exposures were evaluated using toxicity values that are specific to absorbed dermal doses. Most oral toxicity values are based on administered doses rather than absorbed doses. Therefore, in accordance with EPA Region 3 (1995) and EPA (1989a and 2001a) guidance, the toxicity values based on administered doses were adjusted before they were used for evaluating absorbed doses.

Dermal RfDs and SFs were obtained from oral RfDs and SFs via the following relationships:

$$RfD_{Adjusted} = RfD_{Oral} * GI_{Oral}$$

$$SF_{Adjusted} = SF_{Oral} / GI_{Oral}$$

where:

GI_{Oral}	=	Gastrointestinal (GI) Absorption Efficiency (EPA, 1995)
RfD_{Oral}	=	Oral Reference Dose (EPA, 2003a; EPA, 2003b; EPA, 1997a; or EPA-NCEA)
SF_{Oral}	=	Oral Slope Factor (EPA, 2003a; EPA, 2003b; EPA, 1997a; or EPA-NCEA)

Dermally adjusted RfDs and SFs for COPCs are presented in Tables 7.26 (RAGs D Table 5.1) and 7.28 (RAGs D Table 6.1), respectively.

7.5.5 Toxicity Criteria for Chromium

The toxicity criteria for hexavalent chromium (Cr^{+6}) were used in this risk assessment because speciation data (i.e., trivalent versus hexavalent) were not available for samples collected in areas/media of concern at the FFTA Site. Hexavalent chromium was considered to be more toxic than trivalent chromium, therefore, this assumption is conservative in nature.

7.5.6 Toxicity Criteria for Mercury

The toxicity criteria for methyl mercury were used in this groundwater risk assessment because data indicating the form of mercury in environmental media was not available at the FFTA Site. Methyl mercury was considered to be more toxic than inorganic mercury, therefore, this assumption is conservative in nature.

7.5.7 Relative Potency Factors for Dioxins/Furans

The toxicity and cancer risk characterization assessment for chlorinated dioxin and furan congeners was performed using toxicity equivalency factors (TEFs) that relate the toxicity of various congeners to the toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) (EPA, 1989). The TEF for an individual PCDD/PCDF congener is defined as the ratio of the oral carcinogenic slope factor for that congener relative to the slope factor for 2,3,7,8-TCDD. The World Health Organization (WHO) TEF values are published in "Toxicity Equivalency Factors (TEFs) for PCBs, PCDDs, PCDFs for Humans and Wildlife", Environmental Health Perspectives, (Volume 106, Number 12, December 1998).

TEFs were used to convert concentrations of dioxin and furan congeners to a toxicity equivalent concentration (TEQ) for each sample, which represents a concentration of 2,3,7,8-TCDD associated with the same overall cancer risk. Only detected congeners were summed within each sample to yield a total TEQ of 2,3,7,8-TCDD concentration for that sample. Because the maximum TEQ of all samples exceeded the RBC criterion for 2,3,7,8-TCDD, dioxin was selected as a COPC. The total TEQs from all samples was used to estimate the 95 percent upper concentration limit (95 % UCL) of the total TEQ, which was then multiplied by the 2,3,7,8-TCDD cancer slope factor to yield an estimated cancer risk from exposures to all chlorinated dioxin and furan congeners present.

7.5.8 Carcinogenicity of PAHs

Carcinogenic PAHs are related by chemical structure. Only benzo(a)pyrene has an EPA-published SF (EPA, 2003a). All other carcinogenic PAHs, except carbazole, have SFs based on their potency relative to benzo(a)pyrene. The relative potency factors (RPF) for carcinogenic PAH COPCs at the FFTA Site were as follows (EPA, 2003):

- Benzo(a)pyrene (RPF = 1.0)
- Benz(a)anthracene (RPF = 0.1)
- Benzo(b)fluoranthene (RPF = 0.1)
- Benzo(k)fluoranthene (RPF = 0.01)
- Chrysene (RPF = 0.001)
- Dibenz(a,h)anthracene (RPF = 1.0)
- Indeno(1,2,3-cd)pyrene (RPF = 0.1)

7.5.9 Constituents for Which EPA Has Not Developed Toxicity Criteria

EPA has not developed toxicity values for all chemicals for use in quantitative risk characterization. These COPCs include benzo(g,h,i)perylene and phenanthrene. Therefore, risks for these chemicals were evaluated using a surrogate (similar PAH), fluoranthene, which is consistent with previous EPA Region 3 risk assessment projects. RBC screening criteria were applied and resulted in neither substance present at a level greater than the screening criterion.

7.6 RISK CHARACTERIZATION

Potential human health risks resulting from the exposures outlined in the preceding sections are characterized on a quantitative and qualitative basis in this section. Quantitative risk estimates were generated based on risk assessment methods outlined in current EPA guidance (EPA, 1989a).

Noncarcinogenic risk estimates were presented in the form of HQs and HIs that are determined through comparison of estimated intakes with published RfDs. Incremental cancer risk estimates were provided in the form of dimensionless probabilities based on SFs.

Estimated human intakes were developed for each of the specific exposure routes discussed in the preceding sections. Both noncarcinogenic and carcinogenic risks were summarized for each exposure route on a series of tables in this section.

7.6.1 Noncarcinogenic Risks

Noncarcinogenic risk was assessed using the concept of HQs and HIs. The HQ is defined as the ratio of the estimated intake and the RfD for a selected chemical of concern, as follows:

$$HQ = \frac{Intake}{RfD}$$

HIs were generated by summing the individual HQs for the COPCs. If the value of the HI exceeds unity (1.0), the potential for noncarcinogenic health risks associated with exposure to that particular chemical mixture cannot be ruled out (EPA, 1986b). In that case, particular attention should be paid to the target organ(s) affected by each chemical because these are generally the organ(s) associated with RfD-derived effects, and results (HIs) for different organs are not truly additive. The HI is not defined as a mathematical prediction of the severity of toxic effects; it is simply a numerical indicator of exceedance of the acceptable threshold for noncarcinogenic effects. Above an HI of 1, toxic effects would not necessarily occur, but can no longer be ruled out.

7.6.2 Carcinogenic Risks

Incremental cancer risk (ICR) estimates were generated for each of the exposure pathways using the estimated intakes and published SFs, as follows:

$$Risk = Intake * SF$$

If the above equation results in a risk greater than 0.01, the following equation was used:

$$Risk = 1 - e^{-(Intake * SF)}$$

The risk determined using these equations is defined as a unitless expression of an individual's increased likelihood of developing cancer as a result of exposure to carcinogenic chemicals. An ICR of 1×10^{-6} indicates that the exposed receptor has a one in a million chance of developing cancer under the defined exposure scenario. Alternatively, such a risk may be interpreted as representing one additional case of cancer in an exposed population of one million persons. The calculated cancer risks should be recognized as upper-limit estimates. SFs are defined as the upper 95 percent confidence limit of a dose-response curve generally derived from animal studies. Actual human risk, while not identifiable, is not expected to exceed the upper limit based on the SFs and may, in fact, be lower.

7.6.3 Comparison of Quantitative Risk Estimates to Benchmark Criteria

In order to interpret the quantitative risks and to aid risk managers in determining the need for remediation at a site, quantitative risk estimates are compared to typical benchmarks.

A HI exceeding unity (1) indicates that there may be potential noncarcinogenic health risks associated with exposure. If a HI exceeds unity, target organ effects from individual COPCs contributing to the risk are considered. Only those chemicals that impact the same target organ(s) or exhibit similar critical effect(s) will be regarded as truly additive. Thus, COPCs contributing to a HI greater than 1 on the basis of a single target organ/effect are considered to be chemicals of concern (COCs) to be addressed in the feasibility study (FS). Individual COCs that exhibit a HI greater than 1 typically require consideration of remediation options such as treatment/disposal, land use restrictions, monitoring of natural attenuation, or other risk management strategies. However, remediation decisions are not made strictly based on HIs but are often modified by such regulatory requirements as MCLs or chemical-specific clean-up goals.

EPA has defined the range of 1×10^{-4} to 1×10^{-6} as the incremental cancer risk (ICR) "target range" such that, when the sum of cancer risks for all COPCs in a given medium is greater than 1×10^{-4} , this generally

indicates that EPA will require consideration of remediation options such as treatment/disposal, land use restrictions, monitoring of natural attenuation, or other risk management strategies. ICRs below 1×10^{-4} normally will not require initiation of remedial efforts for a given medium. Whenever the overall ICR for a medium is greater than 1×10^{-4} , individual chemicals are selected which contribute significantly to overall risk, typically those chemicals with an individual ICR greater than 1×10^{-6} , and are included in the list of COCs to be addressed in the feasibility study (FS). However, remediation decisions are not made strictly based on ICRs but are often modified by such regulatory requirements as MCLs or chemical-specific clean-up goals.

Potential RME hazard indices and RME cancer risks were estimated for current and future potential receptors. The following sections present a summary of the results of the estimation of risk at areas/media of concern at the FFTA Site.

Receptor risks are presented for each media of concern in the form of tables and summary text. Each of these sections includes summaries of risks estimated by the exposure scenarios. It should be noted that, in each risk summary table where HQs are reported as "N/A", the HQs were not calculable because no RfD has been established. Usually in such cases, carcinogenicity is considered to be more important, since carcinogenicity will generally be seen at lower doses than noncarcinogenic effects. Cancer risks that are reported as "N/A" generally indicate that the chemical is not carcinogenic or that an SF has not yet been developed.

7.6.4 Site-Specific Noncarcinogenic Risks

Site-specific noncarcinogenic risks were estimated for potential receptors at the FFTA Site and are discussed below. If the RME HI did not exceed 1.0 for any target organ group, then the CTE risk was not calculated for that medium and that receptor. Chemical-specific HQs, associated target organs, and the cumulative HI for each target organ are presented on tables in this section according to RAGS D Table 9 format. Supporting data including non-cancer risks, receptor dose, and EPCs for each COPC are presented on tables in Appendix G for each receptor and exposure medium according to RAGS D Table 7 format.

7.6.4.1 Surface Soil

Noncarcinogenic risks did not exceed a HI of 1.0 for any target organ group for the receptors potentially exposed to surface soils at FFTA Site, including the industrial worker, residential child, and residential adult (Tables 7.30, 7.31, and 7.32, respectively).

7.6.4.2 Soil*

Noncarcinogenic risks did not exceed a HI of 1.0 for any target organ group for the industrial worker, construction worker, residential child, and residential adult potentially exposed to soil* (mixture of surface and subsurface soil) at FFTA Site (Tables 7.34, 7.35, 7.36, and 7.37, respectively).

7.6.4.3 Groundwater

Noncarcinogenic risks did not exceed a HI of 1.0 for any target organ group for the construction worker potentially exposed to groundwater at FFTA Site (Table 7.39).

The estimated RME HI for a future residential child exposed to groundwater at FFTA was 38.3 (Table 7.40), exposure contributions primarily from ingestion (and to a lesser extent, dermal absorption), which exceeded the acceptable level of 1.0. The target organs with HIs exceeding 1.0 were CNS (HI of 18.9 – primary contributors manganese and 4-methylphenol); blood (HI of 16 - primary contributors iron and cis-1,2-dichloroethene); liver (HI of 12.9 – primary contributor iron); GI Tract (HI of 12.7 – primary contributor iron); skin and vascular system (HI of 2.7 – primary contributor arsenic); and respiratory tract (HI of 2.3 – primary contributor 4-methylphenol).

The estimated CTE HI for a future residential child exposed to groundwater at FFTA was 21.5 (Table 7.41), exposure contributions primarily from ingestion (and to a lesser extent, dermal absorption), which exceeded the acceptable level of 1.0. The target organs with HIs exceeding 1.0 were CNS (HI of 10.4 – primary contributors manganese and 4-methylphenol); blood (HI of 9.1 - primary contributors iron and cis-1,2-dichloroethene); liver (HI of 7.4 – primary contributor iron); GI Tract (HI of 7.3 – primary contributor iron); skin and vascular system (HI of 1.5 – primary contributor arsenic); and respiratory tract (HI of 1.3 – primary contributor 4-methylphenol).

The estimated RME HI for a future residential adult exposed to groundwater at FFTA was 13.7 (Table 7.42), exposure contributions from ingestion, dermal absorption, and inhalation of vapors during showering, which exceeded the acceptable level of 1.0. The target organs with HIs exceeding 1.0 were CNS (HI of 5.7 – primary contributors manganese and 4-methylphenol via ingestion); blood (HI of 5.1 – primary contributors iron and cis-1,2-dichloroethene via ingestion); liver (HI of 4.1 – primary contributor iron via ingestion); GI tract (HI of 4.0 – primary contributor iron via ingestion); and respiratory tract (HI of 1.8 – primary contributors naphthalene via inhalation and 4-methylphenol via ingestion).

The estimated CTE HI for a future residential adult exposed to groundwater at FFTA was 9.2 (Table 7.43), exposure contributions from ingestion, dermal absorption, and inhalation of vapors during

showering, which exceeded the acceptable level of 1.0. The target organs with HIs exceeding 1.0 were CNS (HI of 4.0 – primary contributors manganese and 4-methylphenol via ingestion); blood (HI of 3.6 – primary contributors iron and cis-1,2-dichloroethene via ingestion); liver (HI of 2.9 – primary contributor iron via ingestion); GI tract (HI of 2.8 – primary contributor iron via ingestion); and respiratory tract (HI of 1.1 – primary contributors naphthalene via inhalation and 4-methylphenol via ingestion).

7.6.5 Site-Specific Cancer Risks

Site-specific cancer risks were estimated for potential receptors at the FFTA Site and are discussed below. If the overall RME ICR did not exceed 1.0×10^{-4} for a given medium and receptor, then the CTE risk was not calculated for that medium/receptor. Chemical-specific ICRs are presented on tables in this section according to RAGS D Table 9 format. Supporting data including cancer risks, receptor dose, and EPCs for each COPC are presented on tables in Appendix G for each receptor and exposure medium according to RAGS D Table 8 format.

7.6.5.1 Surface Soil

The estimated RME ICR for a current/future adult industrial worker exposed to surface soil at FFTA Site was 3.5×10^{-6} (see Table 7.30 for contributions from ingestion, dermal contact, and inhalation of particulates). The ICR is near the lower end of EPA's target risk range of 1×10^{-4} to 1×10^{-6} . The primary contributor to the cancer risk was benzo(a)pyrene via ingestion and dermal contact.

The estimated RME ICR for a future child resident exposed to surface soil at FFTA Site was 1.3×10^{-5} (see Table 7.31 for contributions from ingestion, dermal contact, and inhalation of particulates). The ICR is within EPA's target risk range of 1×10^{-4} to 1×10^{-6} . The primary contributors to the cancer risk were benzo(a)pyrene, dibenz(a,h)anthracene, benzo(b)fluoranthene, and total 2,3,7,8-TCDD toxicity equivalents via ingestion and dermal contact. Dibenz(a,h)anthracene was only detected in four percent of surface soil samples.

The estimated RME ICR for a future adult resident exposed to surface soil at FFTA Site was 6.1×10^{-6} (see Table 7.32 for contributions from ingestion, dermal contact, and inhalation of particulates). The ICR is within EPA's target risk range of 1×10^{-4} to 1×10^{-6} . The primary contributor to the cancer risk was benzo(a)pyrene via ingestion and dermal contact.

The estimated RME incremental cancer risk (ICR) for a future lifetime resident exposed to surface soil at FFTA Site was 2.0×10^{-5} (see Table 7.33 for contributions from ingestion, dermal contact and inhalation of particulates). The ICR is within EPA's target risk range of 1×10^{-4} to 1×10^{-6} . The primary contributors to

the cancer risk were benzo(a)pyrene, dibenz(a,h)anthracene, benzo(b)fluoranthene, benzo(a)anthracene, and total 2,3,7,8-TCDD toxicity equivalents via ingestion and dermal contact. Dibenz(a,h)anthracene was only detected in four percent of surface soil samples.

7.6.5.2 Soil*

The estimated RME ICR for a current/future adult industrial worker exposed to soil* at FFTA Site was 2.8×10^{-6} (see Table 7.34 for contributions from ingestion, dermal contact, and inhalation of particulates). The ICR is near the lower end of EPA's target risk range of 1×10^{-4} to 1×10^{-6} . The primary contributor to the cancer risk was benzo(a)pyrene via ingestion and dermal contact.

The estimated RME ICR for a future adult construction worker exposed to soil* (mixture of surface and subsurface soil) at FFTA Site was less than the lower end of EPA's target risk range of 1×10^{-4} to 1×10^{-6} (see Table 7.35).

The estimated RME ICR for a future child resident exposed to soil* at FFTA Site was 1.08×10^{-5} (see Table 7.36 for contributions from ingestion, dermal contact, and inhalation of particulates). The ICR is within EPA's target risk range of 1×10^{-4} to 1×10^{-6} . The primary contributors to the cancer risk were benzo(a)pyrene, dibenz(a,h)anthracene, and total 2,3,7,8-TCDD toxicity equivalents via ingestion and dermal contact. Dibenz(a,h)anthracene was only detected in one percent of soil* samples.

The estimated RME ICR for a future adult resident exposed to soil* at FFTA Site was 5.0×10^{-6} (see Table 7.37 for contributions from ingestion, dermal contact, and inhalation of particulates). The ICR is within EPA's target risk range of 1×10^{-4} to 1×10^{-6} . The primary contributors to the cancer risk were benzo(a)pyrene and dibenz(a,h)anthracene via ingestion and dermal contact. Dibenz(a,h)anthracene was only detected in one percent of soil* samples.

The estimated RME incremental cancer risk (ICR) for a future lifetime resident exposed to soil* at FFTA Site was 1.6×10^{-5} (see Table 7.38 for contributions from ingestion, dermal contact and inhalation of particulates). The ICR is within EPA's target risk range of 1×10^{-4} to 1×10^{-6} . The primary contributors to the cancer risk were benzo(a)pyrene, benzo(b)fluoranthene, dibenz(a,h)anthracene, and total 2,3,7,8-TCDD toxicity equivalents via ingestion and dermal contact. Dibenz(a,h)anthracene was only detected in one percent of soil* samples.

7.6.5.3 Groundwater

The estimated RME ICR for a future adult construction worker exposed to groundwater at FFTA Site was below EPA's target risk range of 1×10^{-4} to 1×10^{-6} (see Table 7.39 for contributions from dermal contact and inhalation of VOCs in shallow groundwater during excavation activities).

The estimated RME ICR for a future child resident exposed to groundwater at FFTA Site was 1.4×10^{-4} (see Table 7.40 for contributions from ingestion and dermal contact). The ICR is at the upper end of EPA's target risk range of 1×10^{-4} to 1×10^{-6} . The principal COPCs and exposure pathways contributing to the significant portion of cancer risks were arsenic via ingestion (ICR = 1.0×10^{-4}), pentachlorophenol via ingestion and dermal contact (ICR = 1.8×10^{-5}), vinyl chloride via ingestion (ICR = 1.1×10^{-5}), benzene via ingestion (ICR = 5.1×10^{-6}), tetrachloroethene via ingestion and dermal contact (ICR = 3.5×10^{-6}), and bis(2-ethylhexyl) phthalate via ingestion and dermal contact (ICR = 1.3×10^{-6}). Pentachlorophenol was only detected in five percent of groundwater samples.

The estimated CTE ICR for a future child resident exposed to groundwater at FFTA Site was 2.7×10^{-5} (see Table 7.41 for contributions from ingestion and dermal contact). The ICR is within EPA's target risk range of 1×10^{-4} to 1×10^{-6} . The principal COPCs and exposure pathways contributing to the significant portion of cancer risks were arsenic via ingestion, pentachlorophenol via dermal contact, vinyl chloride via ingestion, and benzene via ingestion.

The estimated RME ICR for a future adult resident exposed to groundwater at FFTA Site was 1.8×10^{-4} (see Table 7.42 for contributions from ingestion, dermal contact, and inhalation of vapors during showering). The ICR slightly exceeds EPA's target risk range of 1×10^{-4} to 1×10^{-6} . The principal COPCs and exposure pathways contributing to the significant portion of cancer risks were arsenic via ingestion (ICR = 1.3×10^{-4}), pentachlorophenol via ingestion and dermal contact (ICR = 2.0×10^{-5}), vinyl chloride via ingestion (ICR = 1.4×10^{-5}), benzene via ingestion and inhalation (ICR = 1.1×10^{-5}), tetrachloroethene via ingestion (ICR = 3.1×10^{-6}), chloroform via inhalation (ICR = 2.4×10^{-6}), and bis(2-ethylhexyl) phthalate via ingestion and dermal contact (ICR = 1.5×10^{-6}). Pentachlorophenol was only detected in five percent of groundwater samples.

The estimated CTE ICR for a future adult resident exposed to groundwater at FFTA Site was 4.9×10^{-5} (see Table 7.43 for contributions from ingestion, dermal contact, and inhalation of vapors during showering). The ICR is within EPA's target risk range of 1×10^{-4} to 1×10^{-6} . The principal COPCs and exposure pathways contributing to the significant portion of cancer risks were arsenic via ingestion, pentachlorophenol via dermal contact, vinyl chloride via ingestion, benzene via ingestion, and tetrachloroethene via ingestion and dermal contact.

The estimated RME incremental cancer risk (ICR) for a future lifetime resident exposed to groundwater at FFTA Site was 3.3×10^{-4} (see Table 7.44 for contributions from ingestion, dermal contact, and inhalation of vapors during showering). The ICR exceeds EPA's target risk range of 1×10^{-4} to 1×10^{-6} . The principal COPCs and exposure pathways contributing to the significant portion of cancer risks were arsenic via ingestion (ICR = 2.4×10^{-4}), pentachlorophenol via ingestion and dermal contact (ICR = 3.8×10^{-5}), vinyl chloride via ingestion (ICR = 2.5×10^{-5}), benzene via ingestion and inhalation (ICR = 1.7×10^{-5}), tetrachloroethene via ingestion and dermal contact (ICR = 7.4×10^{-6}), chloroform via inhalation (ICR = 2.4×10^{-6}), and bis(2-ethylhexyl) phthalate via ingestion and dermal contact (ICR = 2.8×10^{-6}). Pentachlorophenol was only detected in five percent of groundwater samples.

The estimated CTE incremental cancer risk (ICR) for a future lifetime resident exposed to groundwater at FFTA Site was 7.6×10^{-5} (see Table 7.45 for contributions from ingestion, dermal contact, and inhalation of vapors during showering). The ICR is within EPA's target risk range of 1×10^{-4} to 1×10^{-6} . The principal COPCs and exposure pathways contributing to the significant portion of cancer risks were arsenic via ingestion, pentachlorophenol via ingestion and dermal contact, vinyl chloride via ingestion, benzene via ingestion, and tetrachloroethene via ingestion.

7.6.6 Blood-Lead Risk Characterization

Blood-lead levels in residential children (age 0 through 6) were estimated using the Integrated Exposure and Uptake Biokinetic (IEUBK) Model (version 1.0) developed by EPA (EPA, 1994b). The model was applied in each applicable medium of concern where lead was selected as a COPC at the FFTA Site.

Lead was selected as a COPC in groundwater but not in surface soil or disturbed soil. Therefore, for the assessment of exposures to lead in groundwater by residential children, the arithmetic mean of the FFTA Site groundwater lead concentration was used (5.64 ug/L), while default values in the IEUBK model were used to represent background lead concentrations in air, house dust, and the level of maternal contribution.

The output of the IEUBK Model is a histogram that presents the estimated percentage of residential children (age 0 through 6 years) with a blood-lead level above 10 ug/dL (considered to be the threshold significance level above which adverse effects cannot be ruled out). When the percentage of the population estimated to have blood-levels above 10 ug/dL is greater than five percent, then EPA considers the potential for adverse effects to be significant (EPA, 1994a).

The estimated percentage of children exposed to groundwater that were predicted to exhibit a blood lead level above 10 ug/dL is 1.2 percent, which is less than EPA's protective level cutoff of 5%. Therefore, no

adverse effects would be anticipated for a hypothetical residential child exposed to lead in groundwater via tap water consumption. Appendix G contains histograms of anticipated blood lead concentrations and tabulated estimates of lead uptake contributions for air, diet, water, soil, and dust for age intervals from 0 through 6 years old.

7.6.7 Summary of Receptor Risks and Hazards for COPCs - Risks Due to Multiple Media

Although carcinogenic and noncarcinogenic risks were summarized separately for each exposure medium in the previous sections, a more realistic scenario is that potential receptors might be exposed to more than one medium, depending upon receptor activity patterns. In this section, additive risks were estimated to show the overall risks that each individual receptor might incur as a result of exposure to multiple media at FFTA Site.

7.6.7.1 Industrial Worker

Based on the receptor exposure pathway assumptions listed in Section 7.4, the current/future industrial worker may be separately exposed to COPCs found in surface soil or soil* (mixture of surface and subsurface soil) but this receptor is not anticipated to be exposed to groundwater. Therefore, it was not necessary to estimate cumulative risks associated with multiple media exposures for this receptor.

7.6.7.2 Construction Worker

Based on the receptor exposure assumptions listed in Section 7.4, the future construction worker may be exposed to COPCs in groundwater and soil* (mixture of surface and subsurface soil). Table 7.46 presents a summation of the RME cancer risks and non-cancer HIs estimated for combined exposure to COPCs in found in these two media. For the construction worker, multiple media exposures are not expected to result in non-cancer HIs exceeding 1.0 for any target organ. In addition, the cumulative cancer risk from multiple media exposure was less than EPA's target risk range of 1×10^{-4} to 1×10^{-6} .

7.6.7.3 Residential Child

Based on the receptor exposure assumptions listed in Section 7.4, the future residential child may be exposed to either groundwater and surface soil or groundwater and soil* (mixture of surface and subsurface soil), depending upon whether soil in a hypothetical future residential neighborhood would remain largely undisturbed or mixed as a result of construction and landscaping. Cancer risks and non-cancer HIs associated with either type of scenario -- combined exposure to COPCs in groundwater and surface soil or combined exposure to groundwater and soil* -- are presented in Tables 7.47 and 7.49,

respectively. For either scenario, multiple media exposures could result in non-cancer HIs exceeding 1.0 for several target organs as a result of contributions from COPCs present in groundwater, whereas contributions to HIs from COPCs in surface soil or soil* were negligible.

For a residential child exposed to groundwater and soil*, the cumulative cancer risk exceeded EPA's target risk range of 1×10^{-4} to 1×10^{-6} , but the relative contributions to the overall cancer risk were much greater in groundwater (greater than 1×10^{-4}), whereas the soil* cancer risk was within the target risk range.

For a residential child exposed to groundwater and surface soil, the cumulative cancer risk exceeded EPA's target risk range of 1×10^{-4} to 1×10^{-6} , but the relative contributions to the overall cancer risk were much greater in groundwater (greater than 1×10^{-4}), whereas the surface soil cancer risk was within the target risk range.

CTE cancer risks and non-cancer HIs for the future residential child exposed to COPCs in multiple media -- groundwater/surface soil and groundwater/soil* -- are presented in Tables 7.48 and 7.50, respectively.

7.6.7.4 Residential Adult

Based on the receptor exposure assumptions listed in Section 7.4, the future residential adult may be exposed to either groundwater and surface soil or groundwater and soil* (mixture of surface and subsurface soil), depending upon whether soil in a hypothetical future residential neighborhood would remain largely undisturbed or mixed as a result of construction and landscaping. Cancer risks and non-cancer HIs associated with either type of scenario -- combined exposure to COPCs in groundwater and surface soil or combined exposure to groundwater and soil* -- are presented in Tables 7.51 and 7.53, respectively. For either scenario, multiple media exposures could result in non-cancer HIs exceeding 1.0 for several target organs as a result of contributions from COPCs present in groundwater, whereas contributions to HIs from COPCs in surface soil or soil* were negligible.

For a residential adult exposed to groundwater and soil*, the cumulative cancer risk exceeded EPA's target risk range of 1×10^{-4} to 1×10^{-6} , but the relative contributions to the overall cancer risk were much greater in groundwater (greater than 1×10^{-4}), whereas the soil* cancer risk was within the target risk range.

For a residential adult exposed to groundwater and surface soil, the cumulative cancer risk also exceeded EPA's target risk range of 1×10^{-4} to 1×10^{-6} , but the relative contributions to the overall cancer risk were

much greater in groundwater (greater than 1×10^{-4}), whereas the surface soil cancer risk was within the target risk range.

CTE cancer risks and non-cancer HIs for the future residential adult exposed to COPCs in multiple media – groundwater/surface soil and groundwater/soil* -- are presented in Tables 7.52 and 7.54, respectively.

7.6.7.5 Lifetime Resident

Based on the receptor exposure assumptions listed in Section 7.4, the future lifetime resident may be exposed to either groundwater and surface soil or groundwater and soil* (mixture of surface and subsurface soil), depending upon whether soil in a hypothetical future residential neighborhood would remain largely undisturbed or mixed as a result of construction and landscaping. Cancer risks associated with either type of scenario -- combined exposure to COPCs in groundwater and surface soil or combined exposure to groundwater and soil* -- are presented in Tables 7.55 and 7.57, respectively.

For a future lifetime resident exposed to groundwater and soil*, the cumulative cancer risk exceeded EPA's target risk range of 1×10^{-4} to 1×10^{-6} , but the relative contributions to the overall cancer risk were much greater in groundwater (greater than 1×10^{-4}), whereas the soil* cancer risk was within the target risk range.

For a future lifetime resident exposed to groundwater and surface soil, the cumulative cancer risk also exceeded EPA's target risk range of 1×10^{-4} to 1×10^{-6} , but the relative contributions to the overall cancer risk were much greater in groundwater (greater than 1×10^{-4}), whereas the surface soil cancer risk was within the target risk range.

CTE cancer risks for the future lifetime resident exposed to COPCs in multiple media – groundwater/surface soil and groundwater/soil* -- are presented in Tables 7.56 and 7.58, respectively.

7.7 UNCERTAINTY ANALYSIS

The goal of the uncertainty analysis is to identify important uncertainties and limitations associated with the groundwater risk assessment. As discussed in EPA (1989), the risk measures used in Superfund site risk assessments are not fully probabilistic estimates of risk but rather are conditional estimates based on a considerable number of assumptions about exposure and toxicity. There are uncertainties associated with each aspect of risk assessment, from environmental data collection through risk characterization.

7.7.1 Uncertainties Associated with Data Collection and Evaluation

Major uncertainties associated with data collection and evaluation are highlighted below.

7.7.1.1 Selection of Locations and Numbers of Samples

The areal extent of the samples (including the number collected and location of the sampling points) in a particular area or medium of interest impacts the selection of COPCs, the calculation of EPCs, and consequently the risks estimated for a site. Generally, sample collection at each area of interest or site should reflect actual site conditions and should include areas that contain the most significant contamination or exposure problems. At the FFTA Site, over 20 monitoring wells encompass the source area on all sides and establish several sampling depth intervals. Older groundwater data were used to identify where data gaps existed and resulted in the installation of additional wells where needed. All groundwater wells were sampled in 2003 within a two month timeframe, which provides a representative snapshot of current groundwater conditions. While the earlier RI investigations did not provide sufficient soil samples to thoroughly characterize the nature and extent of contamination, the supplemental soil samples collected in this investigation provided better coverage throughout the source area, in addition to indicating areas near the boundaries of site contamination. Therefore, the objectives of the Supplemental RI were met in terms of obtaining adequate numbers and locations of samples to fill in data gaps.

Surface water samples were not necessary to collect for the reasons noted in Section 7.4.

7.7.1.2 Data Collection Impacts on Selection of COPCs and Risks

While the extensive additional sampling coverage in the Supplemental RI made it unlikely that the areas of highest contamination were missed, a different potential problem could occur if too few samples were collected as background samples. There were adequate numbers of background soil samples taken to achieve statistical power in comparing site to background, but this was not the case for groundwater because of the requirement that background groundwater samples must be obtained from wells located immediately upgradient of the site rather than just anywhere in the regional area where contamination due to site-related activities was not plausible. There were only 6 upgradient groundwater samples collected in 2003, which was inadequate to allow background tests to confidently demonstrate evidence of metals that should be eliminated as COPCs because of similarity to background levels. Therefore, only a qualitative comparison of site versus background could be performed for metals in groundwater.

Metals found in groundwater that are not actually site-related could be selected as COPCs and site-related human health risks could be overestimated if any of the metals which are risk drivers (namely

arsenic, iron, or manganese) are attributable to background conditions. However, a qualitative comparison of site versus background data suggests the highest observed levels of these metals could be related to site conditions because maximum groundwater concentrations were found within a few dozen feet of the center of the FFTA, in the vicinity where soil samples revealed the greatest impacts of organic contamination. This coincidence potentially indicates either that the risk driver metals were associated with FFTA site activities or that near the center of the FFTA Site reducing conditions related to contaminant degradation have enhanced the dissolution of naturally occurring metals from the geologic formation.

7.7.1.3 Uncertainties Regarding the Selection of the EPC

Other uncertainties exist regarding selection of a concentration for input into the quantitative risk assessment. The use of the exposure point concentration to estimate risk is generally regarded as a conservative estimate since this entails using either the upper 95 percent confidence limit on the arithmetic mean (based on normal or log-transformed data distribution) or the maximum concentration. The choice of the exposure point concentration as the value for input into the risk assessment generally lowers the chances of underestimation of the actual risk present in a pathway at a particular area of interest to a potential receptor.

While the use of upper 95 percent statistical confidence limit on the mean may overestimate the average exposure point concentration, a decision scheme recommended in recent guidance (EPA, 2002b) was followed that reduced the degree of overestimation. Chebychev UCL estimates were generated as appropriate to each situation where the number of samples was small, the log-standard deviation was large, or when the data were non-parametrically distributed. In such cases, the traditional approach would have employed the H-distribution and would have generated unrealistically high UCLs greater than the maximum detected concentration. The newer approach successfully produced more reasonable estimates that were less than the maximum detected value in most cases.

The ability (power) of the W test to be able to correctly identify genuine differences between the shape of a sample population versus a reference normal or lognormal population is reduced when too few samples are collected. If an incorrect distributional assumption is made based on this test, this could lead to an over- or underestimate of the upper 95 percent concentration, which in turn would create some additional uncertainty as to whether the calculated risk is a reasonable approximation of high end exposure. If fewer than 5 samples were available, the distributional tests were not considered significant and the exposure point concentration was selected as the maximum value. The supplemental RI included many samples of each medium, so this situation only occurred rarely, for example, when most values were qualified due to blank contamination or rejected. Aluminum in groundwater and methylene chloride in

surface soil were two analyte data sets for which there were less than 5 data points, but the resulting uncertainty did not significantly impact risk uncertainty because these chemicals were not risk drivers.

Specific uncertainties are associated with the trace (near quantitation limit) concentrations and low frequencies of detection for several compounds that were contributors to cancer risks in groundwater, in particular, with the detection frequencies of 1/19 (5 percent) for pentachlorophenol, 2/21 (10 percent) for vinyl chloride, and 2/21 (ten percent) for tetrachloroethene. Because UCL calculations assumed all non-detected values were positives of magnitude equal to one-half the sample quantitation limit, the resulting 95 percent UCL estimates that were adopted as the EPCs for vinyl chloride and tetrachloroethene could be overestimated and the true mean of the groundwater concentrations might be much lower than these EPCs would suggest. On the other hand, the EPC for pentachlorophenol was based on the maximum value as opposed to a statistical UCL, but use of the maximum represents another large source of uncertainty because only one data point is being relied upon to generate an estimate of the overall potential exposure concentration. (In the case of 4-methylphenol, which was also infrequently detected in 2/19 samples, the two positive results were considerably greater than the quantitation limit, so the impact on the accuracy of the EPC by including one-half the non-detect values is not as severe.) Cancer risks associated with future exposure to groundwater may be overestimated across the majority of the FFTA site for these compounds.

Similar uncertainties are also associated with the low frequencies of detection for Aroclor-1260 and dibenz(a,h)anthracene in surface soil and soil*. For each compound, there was only one positive hit and the detection frequency ranged from one percent to five percent. This caused the EPCs for these compounds in surface soil and soil* to be based on the maximum value as opposed to a statistical UCL, which represents a large source of uncertainty.

7.7.1.4 Uncertainties in Laboratory Data Quality

Validated laboratory data were used to calculate EPCs for groundwater, surface soil, and soil* (mixture of surface and subsurface soil). Established data validation procedures were applied to define analytical uncertainties in terms of qualifying data as inaccurate or imprecise and to eliminate data points that are unusable for risk assessment. This treatment does not eliminate all uncertainty but focuses attention on potential areas of concern regarding accuracy, precision, and data gaps.

7.7.1.5 Uncertainties in Risk-Based Screening Levels

The use of single-route (ingestion) risk-based screening concentrations (RBCs) may lead to the underestimation of risks since these values do not account for the additive effects across various

exposure pathways. In particular, the dermal contact and inhalation pathways are not quantitatively accounted for in the benchmark RBCs used to select or eliminate COPCs in the risk assessment. However, inhalation of outdoor air containing concentrations of organics volatilizing from groundwater or particulate dusts generated from windblown soils are each considered very minor contributions to risk compared to the ingestion pathway.

However, the potential inaccuracy from using ingestion-based RBCs to extrapolate exposures from inhalation of VOCs during showering or dermal contact exposures is somewhat greater. Showering exposures are accounted for via a factor of two adjustments in the Region 3 RBCs, which is generally protective for most, but not all cases. In addition, it is not necessarily true that RBCs based on threshold risks due to ingestion are protective for all dermal contact exposures because of the broad range of variation in GI absorption factors, which alters the effective dermal toxicity factor, and permeability constants, which alter the effective dermal dose. However, the likelihood of underestimating the number of potentially significant COPCs based on using ingestion-based RBCs is mitigated in most cases because Region 3 adjusts the RBC screening benchmarks to a conservative endpoint derived from a target HI of 0.1 for noncarcinogens and a target risk of 1×10^{-6} for carcinogens, which incorporates one or two orders of magnitude, respectively, as a safety factor into the threshold risks associated with the adjusted RBCs.

Benzo(g,h,i)perylene and phenanthrene were detected in soil. Although published RfDs and RBCs were not available for these substances, surrogate screening RBCs were adopted using fluoranthene, a similar PAH compound associated with published RfD and RBC values. Phenanthrene and benzo(g,h,i)perylene were both found to be present below the derived screening RBCs and so were not selected as a COPCs. Accounting for potential noncancer risks in this manner reduces the chance of underestimating risks from compounds without published toxicity factors.

7.7.2 Uncertainties Associated with Exposure Assessment

Major uncertainties associated with Exposure Assessment are listed below.

7.7.2.1 Exposure Scenarios

The likelihood of the occurrence of the defined exposure scenarios is a source of uncertainty. The future anticipated land use at FFTA Site is expected to remain as airfield maintenance and light industrial. Likely potential receptors under this land use scenario include industrial workers or construction workers. Additionally, exposures to groundwater, soil*, and surface soil were evaluated for a hypothetical child and adult resident. This was done in order to generate an upper bound to risks from all potential uses of

groundwater that is consistent with EPA and Commonwealth of Virginia standard guidelines. The degree of exposure to contaminants from the residential scenario would be greater than with other exposure scenarios; however, the continued use of the land as an airfield is anticipated and the development of the area for residential purposes is extremely unlikely.

By assessing the potential risks under a hypothetical residential exposure scenario, the risk assessment can be used to generate standards for protectiveness that are assured to encompass the full range of possible exposures that might occur at the site. However, standards for protectiveness which are based upon residential scenarios would be overly conservative if applied only to prevent adverse health risks in the situation of industrial and construction exposure scenarios. In addition, the assumption that shallow FFTA groundwater (Columbia Aquifer) would be developed as a tap water source for future residences is unlikely because the lower Yorktown aquifer is more productive and use of the shallow Columbia Aquifer is discouraged and restricted by the Commonwealth of Virginia under a groundwater management program administered by the VADEQ.

The future exposure scenario contains an assumption that soils at depths less than 15 feet could eventually become a source of routine human exposure. While future construction of roads, landscaping of development areas, or excavation of typical basements for buildings and residences might result in mixing and redistribution of shallow subsurface soils, disturbance of soil below the 10 or 15-foot depth of a typical basement excavation would be improbable and hence VADEQ guidelines recommend evaluating risks associated with direct contact soil exposure only for samples from depths less than 15 feet. Therefore, it was appropriate and realistic to exclude six subsurface soil samples that were collected at depths below 15 feet from the risk assessment. However, if the six omitted samples were to be included in the total soils data set, then lifetime cancer risks would be 1.6×10^{-4} , which is slightly greater than the upper limit of the target risk range. One particular soil sample (FTA-SU-101-1617) from below the cutoff depth contained concentrations of PAHs notably greater than levels in all other samples.

7.7.2.2 Exposure Equations

There are limitations to using various models and equations to estimate exposure doses or contaminant concentrations. For example, the use of modeled concentrations (i.e., generated fugitive dust concentrations and generated volatile emissions) in place of monitored values may not be indicative of actual site conditions during exposure. Exposure to fugitive dust conservatively assumes that potential receptors will be exposed to the same concentration indoors as outdoors (a very conservative assumption), that soils within an area have unlimited erosion potential, that emissions can be estimated from mean annual wind speed and vegetative cover, and that dispersion concentrations can be estimated from source area, downwind distance to receptors, and region-wide meteorological factors. These

uncertainties were partially offset by the calculation of a site-specific particulate emission factor (PEF) using defined site characteristics and assumptions provided in EPA's Soil Screening Guidance (EPA, 1996c). The effect of the uncertainties using the fugitive dust model is not expected to be significant, because carcinogenic and noncarcinogenic risks estimated using the fugitive dust model were several orders of magnitude below levels of concern in the media of interest.

Exposure by a construction worker to volatile airborne emissions from pooled groundwater during excavation activities conservatively assumes that potential receptors will be exposed to the same vapor concentration in the outdoor breathing zone as at the water surface (a very conservative assumption). Other uncertainties include medium-specific parameters as well as site-specific. There are uncertainties in the physical modeling assumptions such as steady-state vapor- and liquid-phase diffusion through the vadose zone, volumetric constants such as fraction organic carbon, soil* porosity, soil bulk density, and fractional volumetric air content and water content in soils. The effect of the uncertainties using the volatile emission model is not expected to be significant, because carcinogenic and noncarcinogenic risks estimated using the volatile airborne emission model were several orders of magnitude below levels of concern at the areas and media of interest.

The model for dermal exposure to soil assumes that only a very thin, constant thickness layer of soil is available for contaminant transfer to the stratum corneum and that a constant amount of contaminant, proportional to the soil concentration, will be absorbed per unit area of skin and per exposure event. However, adherence to skin varies with such factors as particle size, soil type, and organic carbon content. As estimated by EPA (1992f), the absorbed dermal dose could vary by as much as a factor of 50 from the model estimates, even assuming that activity patterns lead to the exposure duration applied in the experimental trials used to develop absorption factors. Because of the lack of reliable data regarding dermal absorption factors, the risk assessment provides default soil absorption factors for all substances except chemicals for which well-documented absorption factors are available (arsenic, 2,3,7,8-TCDD, PCBs, and PAHs). Even so, considerable uncertainty exists with the accuracy of estimates applied for these substances.

The chemical-specific parameters such as K_{oc} were literature-derived values that are measured under conditions that may or may not be representative of on-site conditions. Parameters such as vapor pressure and solubility were not always obtainable at the desired temperature. Parameters such as the K_d for metals are strongly influenced by other factors such as the chemical form of the metal, the soil type, and pH.

Prediction of absorption rates for lipophilic compounds is difficult due to, among other reasons, the possibility of a second absorption pathway that depends on the lipid content of the stratum corneum at the application site. Experimental determination of absorption rates indicates that interspecies

differences are considerable, which, along with other variability's related to condition and age of skin, differences in lag time, and site of application effects, yields appreciable uncertainty in estimated dermal exposures by using published chemical-specific permeation functions. In addition, literature data indicate a variation by as much as a factor of 300 in chemical absorption rates for skin in different anatomical areas of the body. It should also be noted that children generally have greater absorption rates than adults and their activity patterns often result in greater soil-to-skin adherence factors.

Uncertainties exist in the exposure model for the inhalation of volatiles during showering such as chemical-specific rates of volatilization, droplet size, and droplet residence time in the shower. Most of the inputs into the models were considered conservative; therefore, the output may overestimate the exposure for this route. While the Foster and Chrostowski shower model estimates were not constrained by Henry's Law principles, estimated vapor concentrations in the shower relative to concentrations in the water were relatively low using the input concentrations obtained from FFTA Site groundwater and do not approach or surpass the Henry's Law equilibrium partition coefficient for any COPC modeled.

Uncertainties associated with the lack of groundwater modeling at the site include the assumption that current conditions are indicative of future concentrations of contaminants. Contaminants may increase (due to migration, sediment loading, or chemical transformation) or decrease (due to migration or transformation) over time and vary from area to area. Soil-to-groundwater pathway screening was conducted (see Section 6.4.1) to determine if any substances might be present at levels sufficient to leach into groundwater. This analysis indicated that no substances are anticipated to leach from soil to produce noteworthy groundwater concentrations in the future. In addition, analytical data are available at the FFTA Site spanning more than 10 years and there are clear indications that longitudinal plume migration is occurring along with some evidence for gradual reductions in groundwater contaminant concentrations.

Exposure assumptions can add uncertainty into the risk assessment process based on input values selected for each exposure route. For example, not all people weigh 70 kilograms, drink 2 liters of water per day, and live at the same residence for 30 years. The rationale for each assumption was provided in each table of input parameters. Receptor characteristics, such as age and body weight, were based on published values. Conservative values (based on reasonable maximum exposure data or professional judgment) were used in combination with average values.

7.7.3 Uncertainties Associated With Toxicity Assessment

7.7.3.1 RfDs and SFs

There is uncertainty associated with the RfDs and SFs. The uncertainty results from the extrapolation of animal data to humans, the extrapolation of carcinogenic effects from the laboratory high-dose to the environmental low-dose scenarios, and interspecies and intraspecies variations in toxicological endpoints caused by chemical exposure. The use of EPA RfD values is generally considered to be conservative because the doses are based on no-effect or lowest-observed-effect levels and then further reduced with uncertainty factors to increase the margin of safety by a factor in the neighborhood of 10 to 1,000 fold.

There are uncertainties regarding nonthreshold (carcinogenic) effects extrapolation, from the high doses administered to laboratory animals to the low doses received under more common human exposure scenarios. Uncertainties due to short-time toxicological study predictions of long-term effects are also present. Additionally, there is considerable interspecies variation in toxicological endpoints used in characterizing potential health effects resulting from exposure to a chemical, and there is considerable variability in sensitivity among individuals of any particular species.

The RfDs and SFs of some chemicals have not been established, and therefore toxicity could not be quantitatively assessed. In most cases, where RfDs were unavailable for carcinogens, the carcinogenic risk is considered to be much more significant since carcinogenic effects usually occur at much lower doses.

The uncertainty associated with the dermal exposure is high because of the derivation of the dermal slope factor and reference dose. The dermal toxicity factors are based on default oral absorption factors. This can result in an overestimation of the toxicity factors. In general, dermal exposures at FFTA Site did not drive the carcinogenic or noncarcinogenic risks; therefore, the effects of this uncertainty are expected to be minimal.

As discussed in Section 7.5, established RfDs have an inherent amount of uncertainty. Uncertainty factors for RfDs used in this risk assessment are presented in Tables 7.26 through 7.29. Some chemical specific uncertainties should be noted as follows:

Although the accepted basis for evaluating risk associated with exposure to arsenic is to assume it is a carcinogen, there is uncertainty whether carcinogenic effects are the primary health effects expected to be manifested upon exposure to arsenic. There is some scientific information to indicate that humans are capable of metabolizing arsenic to expedite its elimination from the body (ATSDR, 2000). (Specifically,

the body methylates the arsenic to form monomethyl arsenic and dimethyl arsenic.) There is a limited capacity for the body to metabolize methylate arsenic, but this limit is generally reached when the body's intake of arsenic approximately exceeds 500 µg/day. Generally, concentrations of arsenic at the areas and media of interest at FFTA Site would be expected to correspond to levels that are well within the body's ability to metabolize arsenic. On the other hand, arsenic has been associated with a variety of cancers in epidemiological studies. This adds to the uncertainty regarding carcinogenic risks associated with arsenic exposure.

In nature, chromium (III) predominates over chromium (VI) (Langård and Norseth 1986). Little chromium (VI) exists in biological materials, except shortly after exposure, because reduction to chromium (III) occurs rapidly. Hexavalent chromium can also be transformed to trivalent chromium. However, hexavalent chromium is more soluble, and chromium in water samples is often found to be hexavalent. A usable database of chromium speciation data was not available for the FFTA site; therefore, in performing RBC screening it was conservatively assumed that chromium is present in the hexavalent form. Since chromium was not selected as a COPC in any medium, there is no expected impact on the risk assessment from this uncertainty.

Incidental ingestion of iron exceeded EPA's hazard quotient threshold of 1.0 under certain exposure pathways and receptors. Currently, no toxicity values for iron are published in IRIS or in HEAST. The oral reference dose used to evaluate exposures to iron was obtained from the current EPA Region III RBC tables. This value is based on an allowable daily intake and not on an adverse effect level. In addition, iron is considered an essential nutrient. Consequently, iron may not warrant serious health concerns.

To minimize uncertainty, toxicity factors (SFs and RfDs) should be obtained only from published sources (IRIS, HEAST, and NCEA) because these sources represent peer-reviewed and nationally accepted estimates. However, in this risk assessment, the SF for tetrachloroethene was adopted from an outside source, California's Environmental Protection Agency (CAL-EPA). Although the toxicity value utilized was recommended in the June 17, 2003 update to EPA's risk-based concentration table, CAL-EPA is not generally considered a nationally accepted source for toxicity values. Since the CAL-EPA SF of 0.54 is 27-fold greater than the value previously accepted by EPA (0.02), this contributes substantial uncertainty to the cancer risk estimates generated for tetrachloroethene.

7.7.4 Uncertainties Associated with Risk Characterization

Major uncertainties associated with risk characterization are listed below.

- ICRs and HIs are summed for all potential COPCs and for all applicable routes of exposure. Summing the risks implies that no antagonistic or synergistic effects exist between chemicals. It also assumes that similar mechanisms of action and metabolism are prevalent. Therefore, the use of this approach may either underestimate or overestimate the risks, depending on the chemical-specific interactions, which cannot be predicted. The direction of the uncertainty cannot be defined, but the methodology used is based on current EPA guidance.
- The chemicals of concern (COCs) and their associated cleanup levels for remedial actions are focused only on site-related substances; however, there is a conceivable potential for adverse effects not only from exposure to site-related substances (i.e., COPCs) but also from risks due to non-site-related background substances. Aluminum, arsenic, and iron concentrations were demonstrated to be statistically not greater than background and were therefore eliminated as COPCs from surface soil and soil*. In addition, statistical tests revealed that vanadium concentrations were not greater than background in soil*. A qualitative screening assessment of risks from exposures to the background levels of metals at the FFTA Site can be performed by comparing their FFTA Site soil EPCs (found in Appendix G, distributional analysis tables) versus the unadjusted RBCs, which are based upon a 10^{-6} cancer risk or a HQ of 1.0. In FFTA surface soil and soil*, aluminum, iron, and vanadium EPCs were each less than one-half their respective RBCs at a HQ of 1.0, while the EPCs for arsenic were less than the RBC at a 10.5 risk level. This indicates that the potential for adverse effects from these background metals is negligible for ingestion exposures, although there is need for considering uncertainty regarding risk from dermal contact with soil because RBCs do not account for this pathway of exposure. For the exposure scenarios described in Section 7.4, dermal contact with relatively dry soil having low soil-to-skin adherence factors (see values presented on RAGS D Table 4s) would yield dermal contact risks for these background metals that are less than corresponding ingestion risks for the same receptors under the same conditions of frequency and duration of exposure. Therefore, in soils at the FFTA Site, there is a low likelihood of any significant contributions from background metals concentrations to overall receptor risks.
- Risks to any individual may also be overestimated by summing multiple assumed exposure pathway risks for any single receptor. Although every effort was made to develop reasonable scenarios, not all individual receptors may be exposed via all pathways considered.
- The IEUBK and the adult lead model account for the multimedia nature of lead exposure, incorporates absorption and pharmacokinetic information, and allows the risk manager to consider the potential distributions of exposure and risk likely to occur at a site (the model goes beyond providing a single point estimate output). Although uncertainties are associated with blood-lead modeling using these models, the uncertainties are considered lower than those that conceivably

would result from similar lead evaluations performed using a traditional toxicity slope-based approach. Important uncertainties and limitations in the use of the IEUBK model are listed below.

- The IEUBK model is predictive of blood lead for residential children in the range of 6 months to 7 years of age, which typically is considered to be a more sensitive subpopulation than adults. The model does not apply to adults in either residential or industrial settings. For this reason, the adult lead model could also be considered, which considers the most sensitive adult subpopulation, a pregnant worker. However, lead was not selected as a COPC in any medium to which industrial workers would be potentially exposed.
- Blood-lead variability is characterized by a single number, the geometric standard deviation, which is set to a default value of 1.6 in the IEUBK model and 1.8 in the adult lead model. This value represents the aggregate uncertainty in all sources of population variability, including biological, uptake, exposure, sampling, and analytical components.
- Child blood-lead level predictions obtained using the IEUBK model reflect only the contributions of sources entered into the model and do not take into account any existing body burden that may be the result of prior exposures or any exposures that may have taken place at alternate locations away from the household or neighborhood level, such as parks or daycare centers.

7.8 SUMMARY AND CONCLUSIONS

Site-specific carcinogenic and noncarcinogenic risks were estimated for potential receptors at the FFTA site. Potential receptor scenarios included current and future industrial workers, future construction workers, and hypothetical future residential users. No change in the future use of the site is likely as it is adjacent to an active runway that is an important part of the future facility plan for the installation. The future residential scenario is highly unlikely not only due to the location of the site but it also assumes that use of the shallow aquifer for domestic purposes. Public water is available at the site and use of the shallow aquifer is highly unlikely. The shallow aquifer is not as productive as the lower aquifer present in the area and the use of the shallow aquifer is restricted by the state. As discussed in Section 2.3.4, groundwater use in the area is controlled and allocated through a permitting program administered by the state under the Virginia Groundwater Management Act. The site area is located within a Critical Groundwater Area as designated by the Commonwealth of Virginia and the use of the shallow aquifer is restricted through the issuance of use permits. The SWCB administers the permit program and consults with the Department of Health and local commissions in evaluating permit requests. The Accomack/Northampton County Water Supply Committee has been established as a local commission to work with and advise the state in evaluating permit applications. This committee meets regularly and has established local restrictions on additional withdrawals of groundwater from the shallow aquifer and

require monitoring programs to ensure that existing groundwater users are not impacted by new deep aquifer withdrawals.

7.8.1 Noncarcinogenic Risks

Noncarcinogenic risks did not exceed an HI of 1.0 for any target organ group for receptors potentially exposed to surface soils or total soils (soils*), including the industrial worker, construction worker, residential child, or residential adult. Similarly, noncarcinogenic risks did not exceed an HI of 1.0 for any target organic group for the construction worker potentially exposed to groundwater.

Target organ group HIs greater than 1.0 were estimated for potential domestic use of the shallow groundwater by hypothetical future child and adult residents. The risk drivers include arsenic, iron, manganese, cis-1,2-dichloroethene, 4-methylphenol, and naphthalene

7.8.2 Carcinogenic Risks

Carcinogenic risks were within or less than the acceptable risk range (1×10^{-4} to 1×10^{-6}) for the current and future industrial worker, future construction worker, future child resident, future adult resident, and future lifetime resident potential exposure to surface soils and total soils (soils*)

Carcinogenic risks associated with potential future construction workers exposed to groundwater were less than the acceptable risk range. The risk for hypothetical future child and adult residents exposed to groundwater for domestic purposes, slightly exceeded the acceptable risk range (1.4×10^{-4} and 1.8×10^{-4} , respectively), which resulted in an unacceptable lifetime estimated carcinogenic risk. The primary risk drivers were arsenic, pentachlorophenol, vinyl chloride, benzene, tetrachloroethene, chloroform, and bis(2-ethylhexyl) phthalate. The most significant risks were associated with the potential ingestion of untreated shallow groundwater and the arsenic levels detected in the groundwater. Risks associated with arsenic were the only risks greater than 1×10^{-4} . Eliminating arsenic, results in an overall risk within the acceptable risk range.

In conclusion, unacceptable carcinogenic and noncarcinogenic risks were associated with only hypothetical future residential use of groundwater.

8.0 ECOLOGICAL RISK ASSESSMENT (ERA)

The goal of this ERA is to determine whether adverse ecological impacts are present as a result of exposure to chemicals released to the environment through historical activities at the FFTA. This ERA contains information that will enable risk managers to conclude that either ecological risks at the site are most likely negligible or that further information is necessary to evaluate potential ecological risks at the site.

The ERA methodology follows guidance presented in the following guidance documents:

- Final Guidelines for Ecological Risk Assessment [United States Environmental Protection Agency (USEPA), 1998].
- Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments (USEPA, 1997a).
- Navy Policy for Conducting Ecological Risk Assessment [Department of Navy (DON), 1999].

Figure 8.1 is the Ecological Risk Assessment Tiered Approach.

This ERA consists of Steps 1, 2, and 3a of the eight steps required by the above guidance documents. The first two steps are the screening-level ecological risk assessment (SERA). In the SERA, conservative exposure estimates are compared to screening-levels and threshold toxicity values. The SERA includes the following considerations:

- Problem formulation
- Ecological effects evaluation
- Exposure estimate
- Risk calculation

Step 3a is the first step of the baseline ecological risk assessment (BERA) and consists of refining the conservative exposure assumptions following Steps 1 and 2. Step 3a is meant to further focus the ERA process on the chemicals of greatest concern at a site and is a valuable step to be used in risk management decision-making. Steps 3b through 7 are conducted as part of the BERA, if there are chemicals of potential concern (COPCs) remaining after the Step 3a refinement. Finally, Step 8, Risk Management, is incorporated throughout the ERA process in cooperation with the VADEQ and resource trustees.

8.1 PROBLEM FORMULATION

Problem formulation is the first phase of an ERA and discusses the goals, breadth, and focus of the assessment. It includes general descriptions of the FFTA with emphasis on the habitats and ecological receptors present. This phase also involves characterization of site-related chemicals, chemical sources, migration routes, and an evaluation of routes of chemical exposure. The assessment and measurement endpoints to be evaluated are also selected. Finally, a conceptual model is developed that describes how chemicals associated with the site in question may come into contact with ecological receptors. The following sections provide more detailed descriptions of the steps completed as part of the problem formulation.

8.1.1 Environmental Setting

The Environmental Resources Document (Occu-Health, Inc., 1999) and the Work Plan (TtNUS, Inc., 2003) present detailed descriptions of the terrestrial and wetland habitats present at the Wallops Flight Facility (WFF) and the FFTA. The following sections summarize that description.

WFF is comprised of the Main Land (ML), Main Base (MB), and Wallops Island (WI). The vegetative zones on the ML and MB are diverse and extensive and proceed east to west from marsh to thicket to forest (Occu-Health, Inc., 1999). Habitats within WFF, including the ML, MB, and WI, include dune systems, island forest, salt marsh, upland grasslands, and forest, providing food, shelter, breeding and nesting grounds to a diverse community of species.

The FFTA is located within the MB area. Although a survey of terrestrial species present on the WFF has never been conducted, documentation is available for Assateague Island. Assateague Island neighbors WFF, has similar habitats and settings, and is believed to contain many of the same species. Survey conducted at Assateague Island have revealed the presence of mammals including white-tailed deer (*Odocoileus virginianus*), meadow voles (*Microtus pensylvanicus*), raccoons (*Procyon lotor*), and grey squirrels (*Sciurus carolinensis*); reptiles/amphibians including fowler's toad (*Bufo woodhoussei*), green tree frog (*Hyla cinerea*), black rat snake (*Elapha obsoleta*), hognose snake (*Heterodon platyrhinos*), and box turtle (*Terrapene caolina*); birds including shorebirds such as sanderling (*Calidris alba*), short-billed dowitcher (*Limnodromus griseus*), and various species of tern (*Sterna sp.*) and gulls (*Larus sp.*), songbirds and other avian species such as mockingbirds (*Mimus sp.*), robins (*Turdus migratorius*), and starlings (*Sturnus vulgaris*); and plants including common terrestrial vegetation and wetland species.

Former Fire Training Area

The environmental setting at the FFTA is similar, in large part, to the setting across the WFF, as presented above. The important habitats surrounding the FFTA are a pine woodland stand north and east of the site and scrub-shrub habitats along dirt and brush piles. The field habitat is mowed, limiting potential use by wildlife. Additionally, the FFTA is bounded on one side by an active runway that further deters wildlife from the area. Several perching bird species were observed during a 1994 site reconnaissance (Metcalf & Eddy, Inc., 1996) in areas of the FFTA. Significant bird activity was also noted in the pine stand north of the FFTA. Wildlife habitats observed during the 1994 site reconnaissance indicate the presence of a large number of mammal species including white-tailed deer (*Odocoileus virginianus*) and raccoon (*Procyon lotor*), although these species were not directly observed. It is also likely that small mammals and birds are present at the FFTA. Habitats of reptiles and amphibians noted for the WFF (see above) were not directly observed; however, the presence of preferred habitats at the FFTA indicate these species may be present. Although threatened and/or endangered species are potentially present in other areas of WFF, threatened and/or endangered species are not believed to be present at the FFTA due to the absence of growing or nesting areas that would support such species.

Surface water in the area of the FFTA is limited to intermittent ponded water formed by surface runoff to low-lying areas surrounding the area after rain events (Metcalf & Eddy, Inc., 1996). These areas are not considered wetland quality and do not provide a sufficient habitat to sustain aquatic life; therefore the surface water pathway is incomplete and is not evaluated in this ERA.

8.1.2 Potential Sources of Contamination/Stressor Characteristics

NASA began using the FFTA for fire fighting training exercises in 1965 and continued using the area until 1987 (Metcalf & Eddy, Inc. 1996). It is reported that fire fighting training was conducted twice a week during this time period and consisted of releasing combustible materials onto the ground or into an open-top tank, shallow pit, or discarded airplane body. These materials were ignited and then the flames were extinguished. The identities of the combustible materials used in the training exercises were never recorded.

Section 5.0 of this document describes in detail the potential sources of contamination and the nature and extent of contamination at the FFTA. In summary, VOCs, SVOCs, PCBs, dioxins, and metals have been identified as chemicals potentially present at elevated levels. Physical and chemical characteristics of contaminants affect their mobility, transport, and bioavailability in the environment. Appendix H.1 presents toxicity profiles for the chemical constituents detected in the surface soil and groundwater samples at the site.

8.1.3 Potential Exposure Pathways

In general, chemicals released from materials dumped in terrestrial areas can initially contaminate surface soils. Natural precipitation can then cause the chemical contaminants to leach downward into subsurface soils and groundwater. Discharge of the groundwater can result in the contamination of surface water and sediment inhabited by aquatic receptors.

Terrestrial ecological receptors such as plants, soil invertebrates, mammals, birds, reptiles, and amphibians can be exposed to contaminated surface soil through direct contact. Mammals, birds, reptiles, and amphibians can also ingest contaminated surface soil and food items in which contaminants have accumulated. Some terrestrial receptors such as burrowing mammals or deep-rooted trees could be exposed to shallow layers of contaminated subsurface soils or to shallow groundwater. Most terrestrial receptors are not substantially exposed to subsurface soils or to groundwater that has not discharged to surface water. The shallow groundwater under the site discharges, during periods of elevated groundwater conditions, to an unnamed tributary to Little Mosquito Creek so aquatic organisms may be exposed to chemicals in the groundwater.

8.1.4 Endpoints

8.1.4.1 Assessment Endpoints

Assessment endpoints are explicit expressions of the environmental value that is to be protected (USEPA, 1997a). The selection of these endpoints is based on the habitats present, the migration pathways of probable contaminants, and the routes that contaminants may take to enter receptors.

The habitats present at the FFTA consist of terrestrial habitats such as forested and open areas with grasses and shrubs. Therefore, the assessment endpoints include the protection of the following groups of receptors from adverse effects of contaminants on their growth, survival, and reproduction:

- Soil invertebrates
- Aquatic organisms
- Terrestrial vegetation
- Herbivorous birds and mammals
- Insectivorous birds and mammals
- Reptiles

The following paragraphs discuss why the assessment endpoints listed above were selected for the ERAs.

Soil Invertebrates - Soil invertebrates include earthworms, the juvenile life stages of many insects, and other small organisms that directly inhabit the surface soil. These organisms are expected to be present in the soil in terrestrial habitats at the site. Soil invertebrates promote plant growth by aiding in the formation of soil and through redistribution and decomposition of organic matter. Soil invertebrates also serve as a food source for many mammals and birds. Contaminants can bioaccumulate from the soil into the tissues of soil invertebrates used as a food source by mammals and birds.

Aquatic Organisms - Aquatic organisms could be exposed to contaminated groundwater that discharges into the surface water of the unnamed tributary to Little Mosquito Creek. Aquatic organisms serve as a food source for other aquatic receptors and other higher trophic level animals. Contaminants can bioaccumulate from the water into the tissues of aquatic organisms.

Terrestrial Vegetation - Terrestrial vegetation at the site consists of herbs (grasses, rushes, forbs, ferns, and other non-woody plants), shrubs, woody vines, and trees. These plants serve as a source of food and shelter for many organisms and help prevent soil erosion and excessive surface runoff. Plants can also bioaccumulate some chemical contaminants from the soil that can then be transferred to organisms that feed on plant tissue.

Herbivorous Birds and Mammals - Herbivorous birds and mammals (animals that consume only plant tissue) forage at the site. Their role in the community is essential because, without them, higher trophic levels could not exist. They may be exposed to and accumulate contaminants that are present in the plants they consume.

Insectivorous Birds and Mammals - Insectivorous birds and mammals (animals that consume insects and other invertebrates) present at WFF are considered first-level carnivores because they serve as a food source for higher-trophic-level carnivores. Insectivorous birds and mammals may be exposed to and accumulate contaminants that are present in the food items they consume.

Reptiles - Reptiles are present in the terrestrial areas at the site. These species feed primarily on invertebrates, plants, and small mammals. Reptiles are exposed to and can accumulate contaminants from the food items they consume or from the surface soil in which they live.

USEPA guidance (USEPA, 1997a) states that “it is not practical or possible to directly evaluate risks to all of the individual components of the ecosystem at a site. Instead, assessment endpoints focus the risk

assessment on particular components of the ecosystem that could be adversely affected by contaminants from the site.” Therefore, the ERA will focus on the endpoints tending to yield the highest risks, which should account for endpoints that have lower risks.

Large carnivorous mammals and birds (e.g., red fox, red-tailed hawk) were not selected as assessment endpoints because the greatest exposure to site contaminants is expected to occur to small mammals and birds that ingest invertebrates and/or plants. That is principally because the site is small (less than 1 acre) compared to the typical home range of carnivorous mammals and birds (several hundred acres). Omnivores were not selected as assessment endpoints because exposure to contaminants in plants is greatest for herbivores and exposure to contaminants in animals is greatest for insectivores. Therefore, omnivores are protected by protecting herbivores and insectivores. Risks to large carnivorous and omnivorous mammals and birds are evaluated in the uncertainty section of this ERA.

8.1.4.2 Measurement Endpoints

Measurement endpoints are estimates of biological impacts (e.g., mortality, growth, reproduction) that are used to evaluate the assessment endpoints. The following measures of effects were used to evaluate the assessment endpoints, where appropriate.

- Mortality, growth, and/or reproduction of plants and soil invertebrates were evaluated by comparing chemical concentrations in surface soil to screening values designed to be protective of ecological receptors.
- Mortality, reproduction, and/or developmental effects to birds and mammals were evaluated by comparing estimated ingested doses of contaminants in surface soil, plants, and/or invertebrates to no-observed-adverse-effects-levels (NOAELs) and lowest-observed-adverse-effects-levels (LOAELs).
- Reptiles were evaluated by evaluating risks to birds and mammals because no methods are available to quantitatively evaluate risks to reptiles.
- Mortality, growth, and/or reproduction of aquatic organisms were evaluated by comparing chemical concentrations in the groundwater to surface water screening levels designed to be protective of ecological receptors.

8.1.4.3 Selection of Receptor Species

Many receptors in the soil environments at the FFTA are adequately described in general categories such as soil invertebrates and vegetation. This is due to the nature of the threshold values, effects values, or criteria that are typically used to characterize risk for such organisms. For vertebrate receptors, selection of a particular species is required so that intake through eating and drinking can be estimated.

Ingestion is the primary route of exposure for most mammals and birds. The selection of species used to represent the receptor groups identified in Section 8.1.4.1 was based on considerations of their preferred habitat, body size, sensitivity, home range, abundance, commercial or sport utilization, legal status, and functional role (e.g., predators). To be conservative, chosen representative species have small home ranges. The availability of exposure parameters such as body mass, feeding rate, and drinking rate may also be a factor in selecting surrogate species. The following surrogate species were used for the food chain modeling, which is discussed in Section 8.3:

- Herbivorous mammal: meadow vole
- Herbivorous bird: bobwhite quail
- Insectivorous mammal: short-tailed shrew
- Insectivorous bird: American robin

Receptor profiles for the above species are presented in Appendix H.2.

8.1.5 Conceptual Site Model

A CSM in ERA problem formulation is a written description of predicted relationships between ecological entities and the stressors to which they may be exposed (USEPA, 1998). The CSM consists of two primary components: predicted relationships among stressor, exposure, and assessment endpoint response and a diagram that illustrates the relationships (USEPA, 1998). Figure 8-2 is a pictorial presentation of these relationships. At the FFTA, the source of the chemicals is from former burning activities conducted on the surface soil. Therefore, the immediate exposure medium is the surface soil. Terrestrial plants, invertebrates, and vertebrates are exposed to the surface soil by direct contact, ingestion of soil and other food items. Additionally, site soil contaminants are transported to air via wind erosion and emission of VOCs. Terrestrial vertebrates are exposed to chemicals found in the air via inhalation. Although this pathway is possible, it is not the most significant pathway and is not evaluated in this ERA. Contaminants in the soil may migrate to groundwater, which can be transported to the surface water and sediment of the unnamed tributary to Little Mosquito Creek. Aquatic organisms are exposed to contaminants in the surface water and sediment by direct contact, and ingestion.

8.2 ECOLOGICAL EFFECTS DATA

The preliminary ecological effects evaluation is an investigation of the relationship between the exposure to a chemical and the potential for adverse effects resulting from exposure. In this step, conservative screening levels for evaluating the toxicity to ecological receptors from the detected levels of chemicals at the sites are compiled.

8.2.1 Terrestrial Plants and Invertebrates

Risks to terrestrial plants and invertebrates resulting from exposure to chemicals were evaluated by comparing the chemical concentrations in the surface soil to screening levels. The Region 3 BTAG values (USEPA, 1995) were used as screening levels for this ERA because they were developed specifically for use within USEPA Region 3 to evaluate potential risks to ecological receptors. The Region 3 BTAG values are considered to be conservative screening levels protective of the most sensitive organisms. These toxicity values are expressed in units of concentration because the media of concern are in intimate contact (direct exposure) with terrestrial plants and invertebrates.

8.2.2 Aquatic Organisms

Risks to aquatic organisms resulting from exposure to chemicals were evaluated by comparing the chemical concentrations in the groundwater to surface water screening levels because the groundwater discharges into the surface water. The Region 3 BTAG values (USEPA, 1995), the National Recommended Water Quality Criteria (USEPA, 2002), NOAA Screening Quick Reference Tables (Buchman, 1999), and Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota (Suter and Tsao, 1996) were used as screening levels for this ERA.

8.2.3 Terrestrial Wildlife

The Region 3 BTAG criteria are not designed to evaluate risks to terrestrial wildlife through the ingestion of soil, plants, and/or invertebrates. Therefore, risks to terrestrial receptors from exposures to chemicals found in the surface soil were determined by estimating the chronic daily intake (CDI) and comparing the CDI to toxicity reference values (TRVs) representing acceptable daily doses in mg/kg-day. The TRVs include NOAELs and LOAELs obtained from wildlife studies presented in Oak Ridge National Laboratory (ORNL) Toxicological Benchmarks for Wildlife (Sample et al., 1996).

Appendix H.3 presents the TRVs for the mammals and birds that were used in the ERA and the sources of the TRVs. When a subchronic study was used to develop the TRV, the final value was multiplied by a

factor of 0.1 to account for uncertainty between subchronic and chronic effects. Also, when only a LOAEL study was available, the LOAEL was multiplied by a factor of 0.1 to obtain the NOAEL TRV.

8.3 EXPOSURE CHARACTERIZATION

In order to determine if a chemical has the potential to cause a risk to an ecological receptor, a chemical concentration or chemical dose was compared to the screening levels/TRVs. These chemical concentrations/doses are termed exposure point concentrations (EPCs). The following paragraphs indicate the EPCs that were used for each set of ecological receptors.

8.3.1 Terrestrial Plants and Invertebrates

Terrestrial soil invertebrates and plants can be exposed to contaminants in the surface soil through direct contact and/or ingestion. As indicated in Section 8.2.1, the screening values for these receptors are in units of chemical concentration in each medium. Therefore, the EPCs used for these receptors were the maximum chemical concentrations in each medium.

8.3.2 Aquatic Organisms

Aquatic organisms can be exposed to contaminants present in groundwater that discharges to the surface water. Exposure can be through direct contact and/or ingestion. The EPCs used for these receptors were the maximum chemical concentrations in the groundwater under the site.

8.3.3 Terrestrial Wildlife

The exposures to chemicals for mammals and birds are more complex than the exposures for plants and soil invertebrates and require evaluation of bioaccumulation of chemicals in various food items. Therefore, the EPCs for terrestrial wildlife were estimated in mg/kg-day using exposure dose equations. EPCs for terrestrial wildlife were only calculated for chemicals identified by USEPA as bioaccumulative (USEPA, 2000).

Total exposure of terrestrial wildlife to COPCs in soil (and associated food items such as plants and invertebrates) were determined for the surrogate wildlife species listed in Section 8.1.4.3 using the following equation:

$$CDI = \frac{[(Cf * IR) + (Cs * Is)] * H}{BW}$$

Where:

CDI	=	Chronic daily intake (mg/kg-day)
Cf	=	Chemical concentration in food - (see discussion below)
Cs	=	Contaminant concentration in surface soil (mg/kg)
IR	=	Food ingestion rate (kg/day)
Is	=	Incidental surface soil ingestion rate (kg/day)
H	=	Portion of food intake from the contaminated area (unitless)
BW	=	Body weight (kg)

Contaminant concentrations in food items were calculated as follows using soil to plant or invertebrate bioaccumulation factors (BAFs) from published sources.

$$Cf = Cs * BAF$$

Where:

Cf	=	Chemical concentration in food (mg/kg)
Cs	=	Contaminant concentration in surface soil (mg/kg)
BAF	=	Biota-soil bioaccumulation factor

Appendix H.1 presents chemical-specific dry weight BAFs that were used in this ERA. The following sources were used to obtain most of the BAFs in this ERA, with additional sources used to develop BAFs for the polyaromatic hydrocarbons (PAHs) as described in Appendix H.1:

- Plant BAFs for Organics: Toxicity and Chemical-Specific Factors Database (ORNL, 2002).
- Plant BAFs for Inorganics: Empirical Model for the Uptake of Inorganic Chemicals from Soil by Plants (ORNL, 1998).
- Soil Invertebrate BAFs: Development and Validation of Bioaccumulation Models for Earthworms (Sample et al., 1998).

The diets of the meadow vole and bobwhite quail were assumed to consist of vegetation, while the diets of the short-tailed shrew and American robin were assumed to consist of soil invertebrates (i.e.,

earthworms). Therefore, chemical concentrations in food items of the meadow vole and bobwhite quail were estimated by multiplying each COPCs surface soil concentration by its associated soil-to-vegetation BAF. The chemical concentrations in food items of the short-tailed shrew and American robin were estimated by multiplying each COPCs surface soil concentration by its associated soil-to-invertebrate BAF.

The food chain model scenarios were calculated using various exposure assumptions and TRVs to present a range of risks. For selecting chemicals as COPCs, the following sets of exposure assumptions were used:

- Maximum soil concentrations
- 90th percentile BAFs and biota-sediment bioaccumulation factors (BSAFs) (or maximum value if a 90th percentile value was not available)
- Conservative receptor body weight and ingestion rates
- Receptors spend 100% of their time at the site

For refining the list of COPCs in Step 3a (see Section 8.5), the following set of exposure assumptions were used:

- 95% upper confidence limit (UCL) soil concentration
- Median BAFs (or average value if a median value was not available or appropriate)
- Average receptor body weight and ingestion rates
- Receptors spend 100% of their time at the site. The ecological effects quotients (EEQs) reflect this assumption; however, receptors' home ranges were qualitatively taken into account during the Step 3a evaluation.

and;

- Average soil concentration
- Median BAFs (or average value if a median value was not available or appropriate)
- Average receptor body weight and ingestion rates
- Receptors spend 100% of their time at the site. The EEQs reflect this assumption; however, receptors' home ranges were qualitatively taken into account during the Step 3a evaluation.

Appendix H.2 presents the derivation of the exposure parameters used for the surrogate species. The final exposure values that were used in the food chain model are summarized in Table 8.1. The exposure

assumptions (i.e., ingestion rate, body weight) were obtained primarily from the Wildlife Exposure Factors Handbook (USEPA, 1993); other sources were used when necessary.

8.4 RISK CHARACTERIZATION

The risk characterization is the final phase of an ERA that compares exposure to ecological effects. It is at this phase that the likelihood of adverse effects occurring as a result of exposure to a stressor is evaluated. An EEQ approach was used to characterize the risk to potential terrestrial receptors. This approach characterizes the potential effects by comparing exposure concentrations with the effects data.

When EEQ values exceed 1.0, it is an indication that ecological receptors are potentially at risk; additional evaluation or data may be necessary to confirm with greater certainty whether ecological receptors are actually at risk, especially because most benchmarks are developed using conservative exposure assumptions and/or studies. The EEQ value should not be construed as being probabilistic; rather, it is a numerical indicator of the extent to which an exposure point concentration exceeds or is less than a benchmark.

EEQs for terrestrial plants and invertebrates were calculated as follows:

$$EEQ = \frac{C_{ss}}{SsSL}$$

where:

- EEQ = Ecological effects quotient (unitless)
- C_{ss} = Contaminant concentration in surface soil ($\mu\text{g}/\text{kg}$ or mg/kg)
- SsSL = Plant or invertebrate surface soil screening level ($\mu\text{g}/\text{kg}$ or mg/kg)

EEQs for aquatic organisms were calculated as follows:

$$EEQ = \frac{C_{GW}}{SWSL}$$

where:

- EEQ = Ecological effects quotient (unitless)
- C_{GW} = Contaminant concentration in groundwater ($\mu\text{g}/\text{L}$)
- SWSL = Surface water screening level ($\mu\text{g}/\text{L}$)

EEQs for the terrestrial wildlife models were calculated as follows:

$$EEQ = \frac{CDI}{TRV}$$

where:

- EEQ = Ecological effects quotient (unitless)
- CDI = Chronic daily intake dose (mg/kg-day)
- TRV = Toxicity reference value (NOAEL or LOAEL) (mg/kg-day)

8.5 SELECTION OF CHEMICALS OF POTENTIAL CONCERN

The final part of the screening evaluation includes the selection of ecological COPCs. Chemicals that are not retained as COPCs are assumed to only cause negligible risk to ecological receptors and are not evaluated further in the ERA. Chemicals retained as COPCs are further evaluated in Step 3a to determine if they are carried through as chemicals of concern (COCs). The ecological COPCs were selected by the following procedures:

- Chemicals with EEQs greater than 1.0 based on screening values were retained as COPCs for further evaluation because they have a potential to cause risk to ecological receptors.
- Chemicals with EEQs greater than 1.0 based on the food chain model using NOAELs were selected as COPCs because they have the potential to cause risks to mammals and birds.
- Chemicals without screening levels are retained as COPCs but are only evaluated qualitatively.
- Calcium, magnesium, potassium, and sodium were not retained as COPCs because they are essential nutrients that can be tolerated by living systems even at high concentrations. No evidence indicates that these chemicals are related to site operations, and they are not considered hazardous chemicals.

8.5.1 Risks to Terrestrial Plants and Invertebrates

Table 8.2 presents the selection of the chemicals retained as ecological COPCs in surface soil. Five VOCs and three SVOCs were retained as COPCs because no screening values are available for these chemicals. Fifteen SVOCs (including total PAHs), Aroclor-1260, and 12 metals were retained as COPCs because their maximum detected concentrations exceed the BTAG screening values.

8.5.2 Risks to Aquatic Organisms

Table 8.3 presents the comparison of the maximum chemical concentrations detected in the groundwater at the center of the FFTA to aquatic life surface water screening levels. Based on this comparison of maximum contaminant levels, several chemicals in the groundwater (2-methylphenol, diethyl phthalate, aluminum, iron, lead, and manganese) had EEQs greater than 1.0. However, to assess if these contaminants were migrating to the surface water at concentrations of potential concern, the concentrations of these chemicals in the monitoring wells that are adjacent to the unnamed tributary were then compared to the screening levels to select the COPCs (see Table 8-4). Aluminum, iron, and manganese had EEQs greater than 1.0 in these wells and were selected as COPCs.

8.5.3 Risks to Terrestrial Wildlife

Of the detected chemicals, 14 SVOCs, Aroclor-1260, two dioxins (TEQ for mammals and birds), and nine metals were carried through the food chain model for terrestrial wildlife because they are considered important bioaccumulative compounds (USEPA, 2000). These chemicals are indicated as "BIO" on Table 8.2. Of these chemicals, two PAHs, Aroclor-1260, TEQ (mammal and bird), and six metals were retained as COPCs because they had NOAEL EEQs greater than 1.0 in at least one of the conservative scenario receptor models (see Table 8.5). Risks to terrestrial wildlife from the other bioaccumulative chemicals were considered negligible because NOAEL EEQs for those chemicals were less than 1.0 in all receptor models (see Table 8.5). Appendix H.4 presents the spreadsheet calculations for each receptor's food chain model.

8.6 STEP 3A - REFINEMENT OF CONSERVATIVE EXPOSURE ASSUMPTIONS

Step 3a consists of refining the conservative exposure assumptions/concentrations when evaluating potential risks to ecological receptors (i.e., plants, invertebrates, and wildlife receptors) and re-evaluating the analytical data using benchmarks that are more appropriate for the assessment endpoints. The objective of the Step 3a evaluation is to further reduce the number of chemicals that are retained as COPCs, if possible, in order to focus any additional efforts on those chemicals causing ecological concern.

The first step in the Step 3a refinement is to remove inorganic chemicals with detected concentrations similar to or below background concentrations from the list of COPCs. Although it is recognized that there are potential risks from these chemicals at background levels, chemicals detected at these low concentrations are not related to site activities and should not be further evaluated in the Step 3a process. Although the same can be said for organic chemicals, they are carried through the Step 3a

process and discussed at the end of the risk characterization to be conservative. Table 8.2 has a column that indicates if the chemical was detected at concentrations statistically greater than background concentrations. Section 5.3 of this report presents the background comparison that was conducted for FFTA.

8.6.1 Risks to Terrestrial Plants and Invertebrates

Table 8.2 lists the COPCs that were retained for the potential to cause risks to terrestrial plants and invertebrates. The COPCs are further examined in the following sections. Note that some chemicals with concentrations below the screening values are shaded black because they are bioaccumulative and were included in the food chain model. Because these chemicals did not exceed the screening value, they are not evaluated in this section of the Step 3A refinement because risks to plants and invertebrates are not expected.

Volatiles

2-Hexanone, acetone, methyl acetate, 1,2,4-trimethylbenzene, and 1,3,5-trimethylbenzene were retained as COPCs because no screening values for these chemicals are available. Generally, VOCs are not considered toxic to plants and invertebrates at low concentrations. This is indicated by the relatively high BTAG screening values for other VOCs (i.e., 100 ug/kg to 300 ug/kg); maximum concentrations of 2-hexanone (6 ug/kg), acetone (20 ug/kg), methyl acetate (11 ug/kg), 1,2,4-trimethylbenzene (6 ug/kg), and 1,3,5-trimethylbenzene (2 ug/kg) are well below the screening values for other VOCs. The VOCs also were generally detected at low frequencies. Therefore, although toxicity information for some of the VOCs is not available, risks to plants and invertebrates from these chemicals is unlikely. The uncertainties associated with not retaining the VOCs because toxicity information is not available are discussed further in the uncertainty analysis.

PAHs

As part of this Step 3A refinement, total PAHs were evaluated in the place of individually detected PAH concentrations (see Table 8.2) in the surface soil at the FFTA. This was done because the toxicity of PAHs has been reported to be additive and several studies have reported toxicity data for total PAHs (Di Toro et al., 2000). Also, note that the BTAG screening level for PAHs is not based on risks to plants or invertebrates, it is based on risks to mice. The following bullets list toxicity data for plants and/or invertebrates for total PAHs:

- The Fish and Wildlife Service developed screening levels for total PAHs for plants and invertebrates of 3.1 and 20.8 mg/kg, respectively (Lingenfelter, 2000).
- In USEPA (1997b), total PAHs in soils were reduced from 1,710 mg/kg to 98 mg/kg after the soil was treated, and earthworm mortality decreased from 100% before treatment to 0% after treatment.
- In Environmental Canada (EC) (1994), total PAHs in soils were reduced from 1,442 mg/kg to 36 mg/kg after the soil was treated, and earthworm mortality decreased from 100% before treatment to 0% after treatment in a 28-day test.

The Fish and Wildlife Service study had the most appropriate value. Although all three studies that evaluated risks to plants and/or invertebrates were based on mortality, the Fish and Wildlife Service values reportedly incorporated a safety factor to account for more sensitive sublethal endpoints. Therefore, the 20.8 mg/kg value developed by the Fish and Wildlife Service is appropriate as a screening value for invertebrates; the value of 3.1 mg/kg is an appropriate screening value for plants. Based on the comparison to these values, there is a potential risk to plants and invertebrates at location FFTA-SS-101, which has a total PAH concentration of 33 mg/kg. No impacts to plants and invertebrates are expected from the concentrations of PAHs in the other samples at the site, since the next greatest detection of 1.59 mg/kg is below all the toxicity levels listed above.

Although the maximum detection exceeds the plant screening level for total PAHs, the PAHs do not appear to be significantly impacting the plant community because the site is vegetated. Also, although the invertebrate screening level for total PAHs is lower than the maximum detection, actual risks to invertebrates are uncertain since other studies indicate that the detected concentrations of PAHs are not toxic to earthworms. Finally, sample location FFTA-SS-101 is located in the middle of the site, and is surrounded within about 25 feet with five soil samples (FFTA-SS-102 through -105, and SS03). PAHs were not detected in two of those samples, and the concentrations in the other samples were 0.053 mg/kg, 1.35 mg/kg, and 1.59 mg/kg, thus the elevated detection of 33 mg/kg is well bounded to a small area (maximum of 50 feet by 50 feet). Therefore, if there are any risks to plants and invertebrates from PAHs in the soil, the risks would be limited to a small area.

Bis(2-ethylhexyl) phthalate, Carbazole, and Dibenzofuran

These other SVOCs were retained as COPCs because no screening values are available for comparison. These chemicals were detected infrequently in the samples (2 of 18 for bis(2-ethylhexyl)phthalate, 1 of 25 each for carbazole and dibenzofuran).

Bis(2-ethylhexyl) phthalate was detected in two of 18 samples collected with a maximum detection of 260 ug/kg at location FTA-SS-118. Although a BTAG screening value is not available for bis(2-ethylhexyl)phthalate, the ORNL benchmark for phytotoxicity is 100,000 ug/kg (Efroymson et al., 1997a). The maximum concentration of bis(2-ethylhexyl) phthalate is below this value and so risks to plants are not likely. Also, an ORNL earthworm benchmark for a similar phthalate (dimethylphthalate) is 200,000 ug/kg (Efroymson et al., 1997b). Therefore, risks to earthworms from levels of bis(2-ethylhexyl)phthalate in the soil are not likely.

Carbazole and dibenzofuran were detected in only one of 25 samples each, with detections of 280 ug/kg and 68 ug/kg, respectively, at location FTA-SS-101. Based on the single detection, potential risks appear to be isolated. A toxicity study was identified in the literature for dibenzofuran. In this study, reproductive effects of the enchytraeid worm *Enchytraeus crypticus* exposed to 8 polycyclic aromatic compounds (PACs), including dibenzofuran, were investigated in an agricultural soil. Concentrations were estimated to give a 10% reduction of reproductive output (EC10 values) in the worms and measured at 36,000 ug/kg for dibenzofuran (Sverdrup et al., 2002). The detected concentration of dibenzofuran is well below this value so risks to invertebrates are not likely. No toxicity data were located to evaluate risks to invertebrates or plants from carbazole. However, based on the infrequent detection, and the fact that the detection is bounded within 25 feet in all directions by samples where carbazole was not detected, overall risks to plants or invertebrates from the site would be low because of the small area.

PCBs

Only one PCB, Aroclor-1260, was detected in surface soils at the FFTA. Aroclor-1260 was detected in only one of 26 surface soil samples and was retained as COPC because the single detected concentration exceeded the BTAG screening value. The reference for the BTAG screening level for PCBs is not presented in the source document, however, the endpoint is listed as plants. ORNL has developed a screening benchmark of 40 mg/kg for soil phytotoxicity. The ORNL benchmarks are generally considered to be very conservative values. Therefore, because the ORNL benchmark for PCBs is well above the detected PCB concentration (0.63 mg/kg), risks to plants are unlikely. Also, the site is vegetated with grass which provides another line of evidence that the plants are not being significantly impacted by site contaminants. Also, based on a review of the literature (see below), it is unlikely that the concentration of Aroclor-1260 is impacting earthworms at FFTA.

- A large and significant decline in total microarthropod abundance was observed between Aroclor-1254 concentrations of 500 and 2,500 mg/kg. Therefore, the toxic threshold for PCBs lies somewhere between those concentrations (Parmelee et al., 1997).

- A 14-day laboratory bioassay with crickets yielded an LC₅₀ value of 1,200 mg/kg for Aroclor-1254 (Paine et al., 1993).
- Rhett et al., (1988) reported LC₅₀s of 530 mg/kg for Aroclor-1254 after 14 days and 366 mg/kg after 28 days for earthworms.
- 144 mg/kg of PCBs (primarily Aroclor 1260) was not toxic to two earthworm species after a 14-day laboratory bioassay, the EC₅₀ for cocoon production was approximately 98 mg/kg, and the NOEC for cocoon production was approximately 72 mg/kg (Meier et al., 1997).

Although most of the studies listed above evaluated risks from Aroclor-1254, the toxicity of Aroclor-1260 to invertebrates is expected to be similar to the toxicity from Aroclor-1254. In addition, the low frequency of detection (4 percent) coupled with the low concentration indicates that PCBs are not widespread across the site and that there is not a hot spot of PCB contamination. Therefore, the concentrations of PCBs in the soil are not posing a risk to plants or invertebrates at the FFTA.

Metals

Eleven metals were retained as COPCs in surface soils because their maximum detected concentrations exceeded the screening values. Maximum concentrations of metals were detected in samples across the FFTA and spatially do not represent a clear pattern of contamination. However, of all the metals retained as COPCs only lead was statistically determined to be above background, but background analyses could not be conducted for antimony, mercury, silver, and thallium.

Antimony was detected in six of eight samples collected with a maximum concentration of 1.5 mg/kg at location FTA-SS-106. A statistical comparison to background concentrations could not be conducted (see Table 8.2), however, two of the detected site concentrations (FTA-SS-105 = 0.53 mg/kg and FTA-SS-106 = 1.5 mg/kg) exceeded the maximum background concentration (0.44 mg/kg). The same two samples also were the only ones that exceeded the BTAG screening level of 0.48 mg/kg. FTA-SS-105 and -106 are located less than 40 feet from each other, and are bounded in most directions (within 40 feet or less) by other samples with lower or non-detects concentrations of antimony. Therefore, the area with exceedances of the BTAG screening level is estimated to be less than 80 by 100 feet. The endpoint for the BTAG screening value is not provided but a common conservative benchmark for plants is the ORNL value for phytotoxicity of 5 mg/kg (Efroymsen et al., 1997b). All site concentrations are less than the ORNL value for phytotoxicity, indicating that risks to plants are likely negligible from exposure to antimony in surface soils at the FFTA. The antimony USEPA Ecological Soil Screening Level (SSL) for

invertebrates is 78 mg/kg (USEPA, 2003a). The maximum detection of antimony in the surface soil is well below this level so risks to soil invertebrates are negligible.

Lead was detected in all 26 samples collected with a maximum concentration of 40.8 mg/kg at location FTA-SS-117. The screening value used in the COPC selection for lead was the BTAG value of 0.01 mg/kg and is based on extreme sensitivity by Japanese quail where a significant reduction in both calcium and egg production are noted. A more appropriate benchmark that is based on risks to terrestrial plants and invertebrates are the USEPA Eco SSLs of 210 mg/kg for plants and 1,700 mg/kg for invertebrates (USEPA, 2003b). All detected concentrations of lead at the FFTA are less than the USEPA Eco SSLs. Therefore, risks to plants and invertebrates from exposure to lead in the surface soil at the FFTA are considered negligible.

Mercury was detected in eight of 18 samples collected with a maximum concentration of 0.079 mg/kg at location FTA-SS-116; three of the detections (FTA-SS-105 = 0.071 mg/kg, FTA-SS-116 = 0.079 mg/kg, and FTA-SS-117 = 0.061 mg/kg) slightly exceeded the BTAG screening level of 0.058 mg/kg. A statistical comparison to background concentrations could not be conducted (see Table 8.2); however, the maximum detected site concentration at location FTA-SS-16 is below the maximum background concentration of 0.21 mg/kg indicating that mercury detections at the FFTA are not elevated as compared to background conditions. Also, the endpoint for the BTAG screening value is not provided. A common benchmark that is based on risks to terrestrial plants and invertebrates is the Canadian soil quality guideline (SQG) of 10 mg/kg [Canadian Council of Ministers of the Environment (CCME), 1997]. All detected concentrations of mercury at the FFTA are less than the Canadian SQG; therefore, risks to terrestrial plants and invertebrates from exposure to mercury in the surface soil at the FFTA are considered negligible. Also, as noted above, based on the comparison to the background data, the mercury in the surface soil does not appear to be related to site activities.

Silver was detected in one of ten samples collected with a concentration of 0.19 mg/kg at location FTA-SS-101. A statistical comparison to background concentrations could not be conducted (see Table 8.2); however, the detected silver concentration was only slightly greater than the maximum background concentration of 0.13 mg/kg and was greater than the BTAG screening level of 0.0098 mg/kg. Sample location FFTA-SS-101 is located in the middle of the site, and is surrounded within about 25 feet with four soil samples (FFTA-SS-102 through -105) that were analyzed for silver. Silver was not detected in any of the four samples so the sole detection of 0.19 mg/kg is well bounded to a small area (maximum of 50 feet by 50 feet). Also, based on data in the literature, it is unlikely that the detected level of silver in the soil sample will be bioavailable or toxic. In the Silver Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review (Eisler, 1996), it is reported that silver in soils is largely immobilized by precipitation to insoluble salts and by complexation or adsorption by organic matter, clays, and manganese oxides. The endpoint

for the BTAG screening value is listed only as “corn” in USEPA (1995). A common conservative benchmark for evaluating risks to plants from silver concentrations is the ORNL value for phytotoxicity of 2 mg/kg; the ORNL screening benchmarks are generally considered to be conservative values. In Ratte (1999), artificial soil spiked with silver showed no effects to earthworms at 62 mg/kg after 28 days, and no effects to earthworms were observed after worms were exposed to soil containing 2,000 mg/kg for 14 days. Therefore, because the detected concentration is below the ORNL plant benchmark and below the no effects levels for earthworms, risks to plants and invertebrates are expected to be negligible.

Thallium was detected in only one of 12 samples collected with a concentration of 0.18 mg/kg at location FTA-SS-115, which was greater than the BTAG screening level of 0.001 mg/kg. Thallium was not detected in the background data set. The endpoint for the BTAG screening value is not provided but it is listed as floral. A common conservative benchmark for evaluating risks to plants is the ORNL value for phytotoxicity of 1 mg/kg, which is much greater than the maximum detection. Because the detected concentration is below the ORNL value for phytotoxicity, risks to plants are expected to be negligible. No data were found to evaluate risks to invertebrates from thallium. The sample with the thallium detection was located adjacent to the taxi-way, furthest away from the suspected area of the former fire pit so it is unlikely to be related to site activities. Thallium was not detected in the surface soil sample located 20 feet to the north (FTA-SS-0114). Although no soil samples were collected to the east or west of FTA-SS-115, it is not likely that elevated concentrations of thallium would occur in those samples because of the low frequency of detection across the site and distance from the former fire pit. Therefore, although toxicity data for invertebrates are not available, the infrequent detection, small spatial area the one detection represents, and the relatively low level of that detection, leads to the conclusion that risks to invertebrates at the site from thallium in the soil are negligible.

8.6.2 Risks to Aquatic Organisms

Three metals (aluminum, iron, and manganese) were retained as COPCs in surface water because they were detected in the filtered samples above the screening level in at least one of the three monitoring wells adjacent to the unnamed tributary (see Table 8.4).

The EEQs for aluminum, iron and manganese, based on maximum contaminant detections, were 7.1, 1.6, and 4.9 respectively. The three perimeter monitoring wells located adjacent to the unnamed tributary are located across an approximately 160-foot area and represent general groundwater conditions in the area. Groundwater is expected to enter the tributary across the entire area and would be mixed as it enters the tributary and would also be diluted by the surface water. Average groundwater concentrations were calculated to estimate the expected mixing effect or total groundwater input to the stream (see Table 8-4) The EEQs for aluminum, iron, and manganese, based on the average groundwater concentration,

were 2.8, 0.7, and 2.8 respectively. These EEQs do not take into consideration the surface water dilution that would occur when the groundwater enters the tributary.

The tributary originates upstream of the study area, to the south near the center of the WFF, and is a permitted discharge point for storm sewers, airplane wash racks, and an oil water separator. The stream flow volume is unknown but the stream flows year-round. As indicated in Section 4.5, groundwater from the FFTA area is expected to discharge to the tributary only during periods of high precipitation and elevated water table conditions. The surface water volume and influence from other sources are also expected to increase during these periods, thereby increasing the dilution factor. It should also be noted that Site 14 (a debris pile/landfill) is located between the FFTA and the perimeter wells used in this evaluation. The influence that Site 14 has on the groundwater conditions is unknown at this time. A sample collected from monitoring well 14-GW1 located outside of the influence of the FFTA and adjacent to the Site 14, contained iron and manganese concentrations of 11,400 ug/l (filtered) and 23 ug/l (filtered), respectively.

Overall, the levels of aluminum, iron, and manganese, detected in groundwater in the perimeter wells indicated a potential risk to aquatic organisms. However, considering the average concentrations and the expected dilution that occurs in the tributary, the risks are expected to be low. In addition, based on the low-level and frequency of detection of the metals in other FFTA wells and the levels detected in samples from wells outside of the influence of the FFTA the source of the metals in groundwater is unknown.

8.6.3 Risks to Terrestrial Wildlife

Section 8.5.2 and Table 8.5 lists the chemicals that were retained as COPCs because they had NOAEL EEQs greater than 1.0 in the conservative food chain model for at least one of the wildlife receptors. Less conservative exposure assumptions were then used to recalculate risks to wildlife as explained in Section 8.3.2. Of the chemicals with EEQs greater than 1.0 using the conservative exposure assumptions, only three chemicals had NOAEL EEQs greater than 1.0 under the average input scenario, using the 95% UCL as the chemical concentration in the soil. The robin had a NOAEL EEQ of 1.97 for mercury, and the shrew had NOAEL EEQs of 1.29 and 8.14 for Aroclor-1260 and dioxin TEQ, respectively (see Table 8.6). The dioxin TEQ for the shrew was the only chemical that exceeded a NOAEL EEQ of 1.0 (with an EEQ of 4.1) under the average input scenario, using the average chemical concentration in the soil (see Table 8.7).

Even for the food chain models calculated using less conservative exposure assumptions, the exposure dose was calculated assuming that receptors spend 100% of their time feeding at the FFTA, because the sampled area of the FFTA is less than 0.7 acres is within the home range of some small mammals and

birds. However, the FFTA is primarily an open field with mowed grass adjacent to a taxiway, with little cover in the immediate area. Therefore, although mammals and bird may feed at the FFTA, it is not as likely that birds and mammals make their home at the site and feed solely at the FFTA.

For mercury, the EEQ of 1.97 for the robin slightly exceeds the threshold of 1.0 for a determination of no risks; EEQs for the shrew were less than 1.0. The EEQ of 1.97 is based only on the NOAEL, not the LOAEL. Because NOAELs represent levels at which effects do not occur, an EEQ greater than 1.0 based on the NOAEL does not indicate that risks are occurring, just that they are possible. The exposure dose for the robin (or other birds) assumes that the birds consume 100 percent of their diet from the FFTA and their diet consists of only earthworms. As discussed above, although the home range of some small birds may be smaller than the site, it is not likely that small birds will obtain all of their food from FFTA. In addition, most birds consume a variety of food items based on availability. For example, in one study, fruit constituted more of the diet in fall and winter (>90 percent) than spring (<10 percent) (Sallabanks and James, 1999). Since most of the site is grass covered, robins will not obtain fruit from the site but will obtain fruit from outside the site boundary. Birds also consume other insects that are not in direct contact with the soil, and the chemical concentrations in the most of the insects are expected to be lower than the concentrations in the earthworms. Therefore, the exposure of robins to mercury at FFTA is expected to be lower than estimated. Mercury was detected in 8 of 18 samples at a maximum detection of 0.079 mg/kg. Although the background comparison was inconclusive, the maximum detection in the site samples was less than the maximum detection in the background samples (0.21 mg/kg). This indicates it is not likely that mercury in the soil samples is related to site activities. Therefore, risks to birds from mercury in the soil are expected to be low and similar to background risks.

Aroclor-1260 was only detected in one of 26 samples at a detection of 630 ug/kg in sample SS01. The UCL (146 ug/kg) is biased high because of the one detected concentration, since the reporting limit for Aroclor-1260 in the other samples was less than 42 ug/kg. Therefore, the average concentration of 41.5, which is still greater than the reporting limit in all of the samples except the maximum detection is a more appropriate value for estimating exposure to terrestrial wildlife. The EEQ for Aroclor-1260 using average exposure factors and the average concentration was less than 1.0, so risk to mammals and birds from Aroclor-1260 in surface soil is low.

For the dioxin TEQ, the EEQ of 8.14 (based on the UCL soil concentration) for the shrew exceeds the threshold of 1.0 for a determination of no risks; EEQs for the robin were less than 1.0. The UCL (0.008 ug/kg) is biased high because of the maximum detection (0.0161 ug/kg), since all of the other detections were below the UCL. Therefore, the average concentration of 0.004 ug/kg, which is still greater than the many of the detections, is a more appropriate value for estimating exposure to terrestrial wildlife. Using this average concentration and average exposure factors, the NOAEL EEQ was 4.1 for the shrew;

LOAEL EEQs for the shrew and NOAEL EEQs for the robin were less than 1.0. Because NOAELs represent levels at which effects do not occur, an EEQ greater than 1.0 based on the NOAEL does not indicate that risks are occurring, it means that risks are possible. Similar to what was discussed for mercury, it is not likely that mammals will obtain all of their food from the FFTA. However, because the concentrations of the dioxin TEQs for mammals and birds were statistically similar to background concentrations, risks from dioxins in the site soil samples are similar risks from background soil samples.

Risks to reptiles were not quantitatively evaluated because exposure factors are not established for most species, and toxicity data are very limited. However, birds and mammals were used as a surrogate to evaluate potential risks to reptiles. As discussed above, risks to birds and/or mammals from mercury, PCBs, and dioxins are low, but possible, and are similar to background risks for mercury and dioxins. The robin and shrew were the wildlife receptors with the possible risks, based on an assumed diet of 100 percent earthworms. However, the exposure to site contaminants for reptiles will be lower than the exposure of birds and mammals for several reasons. Snakes and turtles consume food items other than earthworms (i.e., plants, insects, small mammals), which are likely to have lower concentrations of contaminants from the soil than earthworms. Also, as presented in USEPA (1993), the metabolic rates and food ingestion rates for reptiles are lower than those for birds and small mammals. Therefore, because the risks to birds and mammals are low, risks to reptiles from contaminants in the soil are unlikely based on their lower exposure to site contaminants.

8.7 UNCERTAINTY ANALYSIS

Uncertainties associated with most steps of an ERA, include selecting endpoints, collecting data, and evaluating toxicity to the receptors. The following sections describe some of the sources of uncertainty that may be associated with the ERA.

8.7.1 Measurement and Assessment Endpoints

Measurement endpoints were used to evaluate the assessment endpoints selected for the ERA. For this ERA, the measurement endpoints are not the same as the assessment endpoints. Therefore, the measures were used to predict effects to the assessment endpoints by selecting surrogate species. For example, a decrease in reproduction of a robin was used to assess a decrease in the reproduction of the song bird population. However, predicting a decrease in reproduction of robin may either underestimate or overestimate reproductive effects on the song bird population because of differences in ingestion rates, toxicity, food preferences, etc. among different bird species.

Several endpoints were not quantitatively evaluated in this ERA. Risks to burrowing animals were not quantitatively evaluated because the methods for quantifying risks to these species have not been well developed. In addition, risks to reptiles were not evaluated because exposure factors are not established for most species and toxicity data is limited. Therefore, potential impacts to these species were not determined. Risks to large carnivorous and omnivorous birds and mammals were not evaluated in this ERA. Only a small portion of the food for higher trophic level species will actually come from the affected site, so it is difficult to determine the actual risks involved. By evaluating the effects to small birds and mammals, the risks to higher trophic level species is taken into account. Because risks to small mammals and birds were expected to be low or negligible, risks to higher trophic level species will be low as well.

Terrestrial vertebrates could be exposed to contaminated surface water and sediment at the unnamed tributary to Little Mosquito Creek by direct contact or through ingestion of aquatic organisms or surface water. However, these receptors were not evaluated because the tributary is located off-site. The risks to these receptors would be negligible because the contaminant levels are so low.

8.7.2 Exposure Characterization

The chemical dose to terrestrial wildlife was calculated using an equation that incorporates ingestion rates, body weights, BAFs, and other exposure factors. These exposure factors are obtained from literature studies or predicted using various equations. Ingestion rates and body weights vary between species, especially between species inhabiting different areas. For example, the food ingestion rate for the robin was reported as 0.89 g/g-day in California and 1.52 g/g-day in Kansas (USEPA, 1993).

Site-specific tissue samples (i.e., plants, invertebrates) were not collected as part of this investigation. Therefore, chemical concentrations in the tissues of these receptors were estimated using various BAFs (i.e., soil-to-invertebrate, soil-to-plant), to obtain exposure point concentrations in food items for food chain models. These values are derived from the literature and most do not account for site-specific factors such as pH, organic carbon, etc. that can affect BAFs. Therefore, there are uncertainties in the estimation of chemical concentrations in prey items used for food chain models.

Another uncertainty in the exposure calculation is the assumption that chemicals in various media are 100 percent bioavailable to the receptors at the detected concentrations. This is unlikely to occur for chemicals in the environment. As is reported in Efroymson et al. (1997), "the concentrations reported are nominal concentrations of a soluble form (i.e., a highly bioavailable form) of the chemical added to soil. Most metals in natural soils and contaminants of waste sites are in poorly available forms." Therefore, it

is likely that most chemicals (especially metals) in environmental media are much less bioavailable than the chemicals in the laboratory tests used to develop some of the screening levels.

There is uncertainty in chemical data collected at the sites. Measured levels of chemicals are only estimates of the true site chemical concentrations. Many samples were deliberately biased toward known or suspected high concentrations so predicted doses to ecological receptors are likely greater than actual doses.

Because some mammals and birds have small home ranges, the food chain risks are calculated assuming that the receptors feed only at the site. This assumption over-estimates risk because it is unlikely that the receptors will obtain all their food from within the site boundaries or from the most contaminated areas of the FFTA.

There is uncertainty in the actual exposure of aquatic receptors to contaminants in the groundwater. The contaminated groundwater discharges to the unnamed tributary, but the contaminant levels were not measured in the surface water. Because of this, the exposure is estimated based on the levels in the monitoring wells adjacent to the tributary.

8.7.3 Ecological Effects Data

There is uncertainty in the ecological benchmark values. Potentially adverse impacts to terrestrial plants and invertebrates from constituents in surface soil are evaluated by comparing the COPC concentration to surface soil screening values. Few studies and little data are available for establishing soil screening values and many are based on the results of only a few studies. Also, there may be situations where the soil screening values are overpredictive of risk if sensitive species used to develop the criteria are not present. In these cases, an effort was made to identify more appropriate values in the Step 3a refinement. Also, at FFTA, some of the chemical concentrations in the surface soil samples are greater than the plant benchmarks. However, most of the habitat at the site is mowed grass. This provides a one line of evidence that the chemical concentrations in the soil are not impacting the plant community at the site.

The NOAELs and LOAELs selected for the surrogate wildlife species were based on species commonly used in laboratory studies (i.e., rats, mice, ducks). There is uncertainty in the application of toxicity data among species because the contaminant may be more or less toxic to the wildlife species in question than it was to the test species.

The toxicity of chemical mixtures is not well understood. The toxicity data used in the ERA to evaluate

risk to ecological receptors is for individual chemicals. Chemical mixtures can affect organisms very differently than individual chemicals because of synergistic or antagonistic effects. Additionally, toxicological data for some COPCs are sparse. Therefore, there is uncertainty in any conclusions involving the potential impacts to ecological receptors from the combined constituents.

8.7.4 Risk Characterization

Unacceptable risks are possible if an EEQ is greater than or equal to 1.0. However, the magnitude of effects to ecological receptors cannot always be inferred based on the magnitude of the EEQ. Rather, an EEQ greater than 1.0 simply indicates that the dose used to derive the TRV was exceeded. Finally, there is uncertainty in how the predicted risks to a species at the site translate into risk to the population in the area as a whole.

Risks to reptiles were not quantitatively evaluated because exposure factors are not established for most species, and toxicity data are very limited. However, birds and mammals were used as a surrogate to evaluate potential risks to reptiles. Predicting risks to birds and mammals may either over- or under-predict the risks to reptiles because of the differences in ingestion rates, toxicity, food preferences, etc. Therefore, there is uncertainty involved in using risks to birds and mammals to evaluate risks to reptiles.

There is uncertainty in the risks to aquatic organisms based on exposure to the groundwater that discharges to the surface water of the unnamed tributary to Little Mosquito Creek because the actual concentrations of these metals in the surface water are not known.

8.8 SUMMARY AND CONCLUSIONS

Several chemicals detected in the surface soil samples at the FFTA were initially retained as COPCs because their chemical concentrations exceeded screening levels, screening levels were not available for those chemicals, or they were bioaccumulative chemicals with EEQs greater than one based on the conservative exposure scenarios. These chemicals were then re-evaluated in Step 3a of this ERA using a variety of factors such as less conservative exposure assumptions, more appropriate toxicity data based on the assessment endpoints, spatial distribution of the chemicals, and other factors to better determine risks to ecological receptors. The ecological endpoints quantitatively evaluated in this ERA were soil invertebrates, plants, and small mammals and birds that consume terrestrial plants and/or invertebrates.

In addition to the above risk-based discussions, chemical concentrations in the site samples were compared to chemical concentrations in background samples to determine if the chemical were related to

site activities and if potential risks from chemical at the site are similar to background risks. Metals whose concentrations were determined to be statistically similar to background concentrations were eliminated from further discussion at the beginning of the Step 3a. Organic chemicals whose concentrations were determined to be statistically similar to background concentrations were still evaluated in Step 3a, however, to be conservative.

8.8.1 Summary of Risks to Plants and Invertebrates

With the exception of a small area (maximum 50 by 50 feet) where risks to plants and invertebrates are possible, risks to plants and invertebrates are expected to be low or negligible. The chemicals were either detected at concentrations below levels where toxic responses are expected to occur, the chemicals were detected infrequently, or the spatial distribution indicated that the potentially impacted area was small. Also, the habitat at the site is primarily a mowed field, which is maintained as such because of the adjacent taxiway. The following paragraphs discuss the risks for each chemical or group of chemicals that were retained as COPCs.

VOCs: Five VOCs were retained as COPCs because screening values for these chemicals were not available. Generally, VOCs are not considered toxic to plants and invertebrates at low concentrations, which are indicated by the relatively high BTAG screening values for other VOCs. The VOCs without screening levels were generally detected at low frequencies and their maximum concentrations were less than the screening values for the other VOCs. Therefore, although toxicity information for some of the VOCs is not available, risks to plants and invertebrates from these chemicals is unlikely.

PAHs: As part of this Step 3A refinement, total PAHs were evaluated in the place of individually detected PAH concentrations in the surface soil because the toxicity of PAHs has been reported to be additive and several studies have reported toxicity data for total PAHs. The maximum total PAH concentration in the site soil samples is greater than a screening level developed in one study for risk to plants and invertebrates, but below the levels that caused toxicity to earthworms in two other studies. Therefore, although it is possible that the concentrations of PAHs in the soil are impacting plants and invertebrates at the location of the maximum detection, impacts to invertebrates appear to be less likely based on the toxicity data. No impacts to plants and invertebrates are expected from the concentrations of PAHs in the other samples at the site based on the low PAH concentrations in the other samples. Although the maximum detection exceeds the plant screening level for total PAHs, the PAHs do not appear to be significantly impacting the plant community since the site is vegetated. Also, although the invertebrate screening level for total PAHs is lower than the maximum detection, actual risks to invertebrates are uncertain since other studies indicate that the detected concentrations of PAHs are not toxic to earthworms. Finally, if there are any risks to plants and invertebrates from PAHs in the soil, the risks

would be limited to a small area (maximum of 50 feet by 50 feet), because the sample with the maximum detection is well bounded by other samples with much lower PAH concentrations.

Other SVOCs: Three additional SVOCs were retained as COPCs because no screening values are available for comparison. These chemicals were detected infrequently in the soil samples (detection frequencies of 2/18 or 1/25). Based on available toxicity data for those SVOCs (or similar SVOCs), risks to plants and invertebrates are not likely. Where toxicity data was lacking, the low frequencies of detection and relatively low concentrations lead to the conclusion that these SVOCs are not impacting plants or invertebrates at the site.

PCBs: Aroclor-1260 was the only PCB was detected in surface soils at the FFTA, and it was only detected in one of 26 surface soil samples. The maximum detection was well below the conservative ORNL benchmark for plants so risks to plants are unlikely. Also, based on a review of the toxicity data for earthworms, it is unlikely that the concentrations of PCBs are impacting earthworms. Finally, the low frequency of detection (4 percent) coupled with the low concentration indicates that PCBs are not widespread across the site and that there is not a hot spot of PCB concentration. Therefore, the concentrations of PCBs in the soil are not posing a risk to plants or invertebrates at the FFTA.

Metals: Eleven metals were retained as COPCs in surface soils because their maximum detected concentrations exceeded the screening values. Lead was the only metal that was statistically determined to be detected at concentrations greater than background, but background analyses could not be conducted for antimony, mercury, silver, and thallium.

Antimony: All site concentrations are less than the ORNL value for plants and the USEPA Eco SSL for invertebrates, indicating that risks to plants and invertebrates are likely negligible from exposure to antimony in surface soils at the FFTA.

Lead: All site concentrations are less than the USEPA Eco SSL for plants and invertebrates, indicating that risks to plants and invertebrates are likely negligible from exposure to lead in surface soils at the FFTA.

Mercury: All site concentrations are less than the Canadian SQG for plants and invertebrates, indicating that risks to plants and invertebrates are likely negligible from exposure to mercury in surface soils at the FFTA. Also, based on the comparison to the background data, the mercury in the surface soil does not appear to be related to site activities.

Silver: The one detected concentration of silver is below the ORNL plant benchmark and below the no effects levels for earthworms indicating that risks to plants and invertebrates are expected to be negligible. Also, one the silver detection was just slightly greater than the background concentration and is well bounded to a small area (maximum of 50 feet by 50 feet).

Thallium: Thallium was detected in only one of 12 samples at a concentration below ORNL benchmark for plants indicating that risks to plants are expected to be negligible. No data was found to evaluate risks to invertebrates from thallium. However, the infrequent detection, small spatial area the one detection represents, and the relatively low level of that detection, leads to the conclusion that risks to invertebrates at the site from thallium in the soil are negligible.

8.8.2 Summary of Risks to Aquatic Organisms

Aluminum, iron, and manganese were retained as COPCs because their maximum detections in the perimeter wells adjacent to the unnamed tributary to Little Mosquito Creek exceeded surface water screening levels. There is a lot of uncertainty in this evaluation because the actual concentrations of these metals in the surface water are not known and the source of the metals is not well defined. Some dilution will occur when the groundwater enters the surface water but the amount of dilution cannot be quantified. Therefore, although risks to aquatic organisms from aluminum, iron, and/or manganese in the groundwater are possible (after it discharges to the surface water), the metals may not be site related and the overall risks are expected to be low.

8.8.3 Summary of Risks to Terrestrial Wildlife

Several chemicals were retained as COPCs because they had NOAEL EEQs greater than 1.0 in the conservative food chain model for at least one of the wildlife receptors. Less conservative exposure assumptions were then used to recalculate risks to wildlife as part of the Step 3a refinement. Only three of the COPCs had NOAEL EEQs greater than 1.0 under the average input scenario, using the 95% UCL as the chemical concentration in the soil: Aroclor-1260, dioxins, and mercury. Dioxins were the only chemicals that exceeded a NOAEL EEQ of 1.0 under the average input scenario, using the average chemical concentration in the soil.

PCBs: Because Aroclor-1260 was only detected in one of 26 samples, and the one detection was well above the reporting limit for the other samples, the UCL was biased high. Therefore, the average concentration, which is still greater than the reporting limit in all of the samples except the maximum detection is a more appropriate value for estimating exposure to terrestrial wildlife. The EEQ for Aroclor-

1260 using average exposure factors and the average concentration was less than 1.0, so risk to mammals and birds from Aroclor-1260 in surface soil is low.

Dioxins: The UCL for the dioxin TEQ also is biased high because of the maximum detection, since all of the other detections were below the UCL. Therefore, the average dioxin TEQ concentration, which is still greater than the many of the detections is a more appropriate value for estimating exposure to terrestrial wildlife. Using that concentration and average exposure factors, the NOAEL EEQ for the shrew was 4.1; LOAEL EEQs for the shrew and NOAEL EEQs for the robin were less than 1.0. Because NOAELs represent levels at which effects do not occur, an EEQ greater than 1.0 based on the NOAEL does not indicate that risks are occurring, it means that risks are possible. However, because the concentrations of the dioxin TEQs for mammals and birds were statistically similar to background concentrations, risks from dioxins in the soil are possible, but are similar to background risks.

Mercury: For mercury, the NOAEL EEQ of 1.97 for the robin; LOAEL EEQs for the robin and NOAEL EEQs for the shrew were less than 1.0. Because NOAELs represent levels at which effects do not occur, an EEQ greater than 1.0 based on the NOAEL does not indicate that risks are occurring, just that they are possible. The exposure of robins to mercury at FFTA is expected to be lower than estimated and similar to the exposure from background concentrations. Therefore, risks to birds from mercury in the soil are possible, but are similar to background risks.

REFERENCES

GENERAL REFERENCES

Brownlee, K.A. 1965. *Statistical Theory and Methodology in Science and Engineering*. New York, New York. John Wiley and Sons, Incorporated.

Cline, Patricia V., and Daniel R. Viste, 1984. *Migration and Degradation Patterns of Volatile Organic Compounds*. Presented at the Fifth National Conference on Management of Uncontrolled Hazardous Waste Sites, Washington, D.C.

Commonwealth of Virginia. 2002. *Water Quality Standards 9VAC25-260-5 et seq.* State Water Control Board. S62.1-44.15(3a) of the Code of Virginia. June 5.

Dragun, James, Ph.D., 1988. *The Soil Chemistry of Hazardous Materials*. Hazardous Materials Control Research Institute, Silver Springs, Maryland.

Ebasco Services, Inc. 1990. *Final Report of Site Investigation for Wallops Flight Facility*. Wallops Island, VA.

Gibbons, J.A., Alexander, M. 1989. *Microbial Degradation of Sparingly Soluble Organic Chemicals: Phthalate Esters*. *Environmental Toxicological Chemistry*. Vol. 8, no. 4. pp. 283-291

Lyman, W.J., W.F. Reehl, and D.W. Rosenblatt, 1990. *Handbook of Chemical Property Estimation Methods*. American Chemical Society, Washington, D.C.

Metcalf and Eddy. 1992. *Final Design Investigation of the Aviation Fuel Tank Farm Area*. June.

Metcalf and Eddy. 1996a. *Site Inspection for Miscellaneous Sites at Wallops Flight Facility*. March.

Metcalf and Eddy. 1996b. *NASA Goddard Space Flight Center, Wallops Flight Facility, Former Fire Training Area Remedial Investigation Final Report*. February.

NASA. 2002 *Wallops Flight Facility Internet Homepage*. www.wff.nasa.gov

Occu-Health, Inc. 1999. *Environmental Resources Document, NASA Goddard Space Flight Center, Wallops Flight Facility, Wallops Island, VA*. October

GENERAL REFERENCES (Continued)

Tetra Tech NUS (TtNUS) 2003a. Work Plan for Supplemental Remedial Investigation Activities at the Former Fire Training Area and Waste Oil Dump, NASA Wallops Flight Facility, Wallops Island, Virginia. January.

Tetra Tech NUS (TtNUS) 2004. Draft Background Soil and Groundwater Investigation Report for the Main Base, NASA Wallops Flight Facility, Wallops Island, Virginia. May.

TOXNET. 2001. Hazardous Substances Data Bank. National Library of Medicine. Specialized Information Services. <http://www.toxnet.nlm.nih.gov>.

United States Army Corps of Engineers. 2000. Wallops Flight Facility-Main Base-GIS Based Historical Photographic Analysis. 1938-1940. November.

United States Department of Agriculture (USDA). 1994. Soil Conservation Service. Soil Survey of Accomack County, VA. September.

United States Environmental Protection Agency, 1979. Water-Related Fate of 129 Priority Pollutants. EPA/440/4-79-029. Monitoring and Data Support Division, Washington, D.C.

United States Environmental Protection Agency, 1982. Aquatic Fate Process Data for Organic Priority Pollutants. EPA/440/4-81-014. Office of Drinking Water Regulations and Standards, Washington, D.C.

United States Environmental Protection Agency. 1994c. Data Quality Objectives Decisions Error Feasibility Trials (DQO/DEFT). EPA 600/R-96.EPA G-4D Office of Research and Development.

United States Environmental Protection Agency. 1994d. Guidance for the Data Quality Objectives Process. EPA QA/G-4. EPA/R-96/055. Office of Research and Development.

United States Environmental Protection Agency. 1996d. Soil Screening Guidance. EPA/540/R-96/018. Office of Emergency and Remedial Response. Washington, DC. April 1996.

United States Geological Survey (USGS) (EPA). 1968

Versar, Inc. 2000. Risk Assessment Update Former Fire Training Area. Goddard Space Flight Center, Wallops Flight Facility, Wallops Island, VA. March.

GENERAL REFERENCES (Continued)

Verschueren, 1983. Handbook of Environmental Data of Organic Chemicals-Second Edition. Van Nostrand-Reinhold Company, New York, NY.

ERA REFERENCES

Buchman, M. F., 1999. NOAA Screening Quick Reference Tables, NOAA HAZMAT Report 99-1, Seattle, WA, Coastal Protection and Restoration Division, National Oceanic and Atmospheric Administration. <http://response.restoration.noaa.gov/cpr/sediment/squirt/squirt.html>

CCME, 1997. Recommended Canadian Soil Quality Guidelines. Canadian Council of Ministers of the Environment. Ottawa, Ontario. March.

Di Toro, D.M., J.A. McGrath, and D.J. Hansen. 2000. Technical Basis for Narcotic Chemicals and Polycyclic Aromatic Hydrocarbon Criteria. I. Water and Tissue. Environmental Toxicology and Chemistry. Vol. 19, No. 8. pp. 1951-1970.

EC (Environmental Canada). 1994. In-Situ, On-Site Bioremediation of Wood Treatment Soils Containing Chlorinated Phenols and PAHs. Grace Dearborn Inc., Mississauga, Ontario. (http://www.ec.gc.ca/desrt/technical/16_e.htm)

Efroymsen, R.A., M.E. Will, and G.W. Suter II. 1997a. Toxicological Benchmarks for Contaminants of Potential Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Process: 1997 Revision. Oak Ridge National Laboratory. November. ES/ER/TM-126/R2.

Efroymsen, R.A., M.E. Will, G.W. Suter II, and A.C. Wooten. 1997b. Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Terrestrial Plants: 1997 Revision. Oak Ridge National Laboratory. November. ES/ER/TM-85/R3.

Eisler, Ronald. 1996. Silver Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. US Department of Interior – Fish and Wildlife Service. Biological Report 32. September.

Lingenfelter, S. 2000. Addendum to the Final Work Plan for the Former Skeet Range (Site 17B), U.S. Army Transportation Center, Fort Eustis, Virginia. U.S. Fish and Wildlife Service. September 7.

Meier, J.R., L.W. Chang, S. Jacobs, J. Torsella, M.C. Mecks, and M.K. Smith, 1997. Use of Plant and Earthworm Bioassays to Evaluate Remediation of Soil from a Site Contaminated with Polychlorinated Biphenyls. Environmental Toxicology and Chemistry, Vol. 16, No. 5, pp. 928-938.

Metcalf and Eddy. 1996. Former Fire Training Area Remedial Investigation Final Report. NASA Goddard Space Flight Center, Wallops Flight Facility, Wallops Island, Va. February.

ERA REFERENCES (Continued)

Navy (Department of the Navy). 1999. Navy Policy For Conducting Ecological Risk Assessments. Memo from Chief of Naval Operations to Commander, Naval Facilities Engineering Command, 05 April 1999. Department of the Navy, Washington, DC.

Occu-Health, Inc. 1999. Environmental Resources Document, NASA Goddard Space Flight Center, Wallops Flight Facility, Wallops Island, Va. October

ORNL (Oak Ridge National Laboratory). 1998. Biota Sediment Accumulation Factors for Invertebrates: Review and recommendations for the Oak Ridge Reservation. BJC/OR-112. August.

ORNL (Oak Ridge National Laboratory), 2002. Toxicity and Chemical-Specific Factors Database. Oak Ridge National Laboratory Web Page, http://risk.lsd.ornl.gov/cgi-bin/tox/TOX_9801.

Paine, J.M., M.J. McKee, and M.E. Ryan. 1993. Toxicity and Bioaccumulation of Soil PCBs in Crickets: Comparison of Laboratory and Field Studies. *Environmental Toxicology and Chemistry*, Vol. 12, pp. 2097-2103.

Parmelee, R.W., C.T. Phillips, R.T., Checkai, and P.J. Bohlen, 1997. Determining the Effects of Pollutants on Soil Faunal Communities and Trophic Structure Using a Refined Microcosm System. *Environmental Toxicology and Chemistry*, Vol. 16, No. 6, pp. 1212-1217.

Ratte H.T., 1999. Bioaccumulation and Toxicity of Silver Compounds: A Review. *Environmental Toxicology and Chemistry*, Vol. 18, No. 1, pp. 89-108.

Rhett, W.N., D.M.M. Adams, P. Roza, and L. Henzen, 1988. Lethal and Sublethal Effects of Aroclor 1254 on *Eisenia foetida*. TNO-report R88/335. Netherlands Organization for Applied Scientific Research, Delft, The Netherlands. Cited in Parmelee et al., 1997.

Sallabanks, R., and F.C. James. 1999. American Robin (*Turdus migratorius*). In *The Birds of North America*, No. 462 (A. Poole and F. Gill, eds.). The Birds of North America, Inc., Philadelphia, PA.

Sample, B.E., D.M. Opresko, and G.W. Suter II. 1996. Toxicological Benchmarks for Wildlife: 1996 Revision. Oak Ridge National Laboratory. June. ES/ER/TM-86/R3.

ERA REFERENCES (Continued)

Sample, B.E., J.J. Beauchamp, R.A. Efroymson, G.W., Suter II, and T.L. Ashwood. 1998. Development and Validation of Bioaccumulation Models for Earthworms. Oak Ridge National Laboratory. June. ES/ER/TM-220.

Suter, G.W. II. and C.L. Tsao. 1996. Toxicological Benchmarks for Screening Potential Constituents of Concern for Effects on Aquatic Biota:1996 Revision. Environmental Sciences Division, Oak Ridge National Laboratory. ES/ER/TM-96/R2.

Sverdrup, L.E., J. Jensen, A.E. Kelley, P.H., Krogh, and J. Stenersen. 2002. Effects of Eight Polycyclic Aromatic Compounds on the Survival and Reproduction of *Enchytraeus Crypticus* (Oligochaeta, Clitellata), *Environmental Toxicology and Chemistry*, Vol. 21, No. 1, pp. 109-114.

Tetra Tech NUS, Inc., 2003. Work Plan for Supplemental Remedial Investigation Activities at the Former Fire Training Area and Waste Oil Dump, NASA Wallops Flight Facility. Prepared for the National Aeronautics and Space Administration, Goddard Space Flight Center. January.

USEPA (U.S. Environmental Protection Agency), 1993. Wildlife Exposure Factors Handbook. Office of Research and Development. Washington, D.C. December 1993. EPA/600/R-93/187a.

USEPA (U.S. Environmental Protection Agency), 1995. Region III BTAG Screening Levels. Region III Biological Technical Assistance Group. Philadelphia, PA. January.

USEPA (U.S. Environmental Protection Agency), 1997a. Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments. Interim Final. Environmental Response Team. June 5.

USEPA (U.S. Environmental Protection Agency), 1997b. GRACE Bioremediation Technologies' DARAMEND Bioremediation Technology. Office of Research and Development. Washington, D.C. EPA/540/R-95/536a.

USEPA (U.S. Environmental Protection Agency), 1998. Final Guidelines for Ecological Risk Assessment. Effective April 30.

USEPA (U.S. Environmental Protection Agency), 2002. National Recommended Water Quality Criteria: 2002. Office of Water. EPA 822-R-02-047. November.

ERA REFERENCES (Continued)

USEPA (U.S. Environmental Protection Agency), 2000. Bioaccumulation Testing and Interpretation for the Purpose of Sediment Quality Assessment, Status and Needs. Office of Water, Office of Solid Waste. EPA 823-R-00-001. February.

USEPA (U.S. Environmental Protection Agency), 2003a. Ecological Soil Screening Level for Antimony, Interim Final. Office of Emergency and Remedial Response. OSWER Directive 9285.7-61. August.

USEPA (U.S. Environmental Protection Agency), 2003b. Ecological Soil Screening Level for Lead, Interim Final. Office of Emergency and Remedial Response. OSWER Directive 9285.7-70. August.

HHRA REFERENCES

Agency for Toxic Substance and Disease Registry (ATSDR). 2000. Toxicological Profile for Arsenic PB/2000/108021. September.

Cowherd, C., G.E. Muleski, P.J. Englehart, and D.A. Gillette. 1984. Rapid Assessment of Exposure to Particulate Emissions from Surface Contamination Sites. Midwest Research Institute, Kansas City, MO.

Foster, S. A. and P.C. Chrostowski. 1987. Inhalation Exposures to Volatile Organic Contaminants in the Shower. Presented at the 80th Annual Meeting of the Air Pollution Control Association. New York, NY. June.

Gehan, E.A., 1965. A Generalized Wilcoxon Test for Comparing Arbitrarily Singly-Censored Samples. *Biometrika*. Volume 52, pp. 203-223.

Gilbert, R.O. 1987. *Statistical Methods for Environmental Pollution Monitoring*. Van Nostrand and Reinhold, New York, NY.

Gilbert, R.O. 1993. Letter to Beverly Ramsey, Systematic Management Services, Incorporated. Battelle Pacific Northwest Laboratories. Richland, Washington. July.

Langard, S., and Norseth, T. 1986 "Chromium", *Handbook on the Toxicology of Metals*, Freiberg, L., Nordberg, G.F., and Vouk, V.B. eds., Vol. II, Elsevier Science Publishers B.V., New York, pp. 185-210

Millard, S.P. and Deverel, S.J., 1988. Nonparametric Statistical Methods for Comparing Two Sites Based on Data With Multiple Nondetect Limits. *Water Resources Research*. Volume 24, No. 12, pp. 2087-2098. December.

Naval Facilities Engineering Command (NFEC). 1998. *Procedural Guidance for Statistically Analyzing Environmental Background Data*. San Diego: Southwest Division, Naval Facilities Engineering Command (SWDIV) and Engineering Field Activity (EFA) West. September.

Naval Facilities Engineering Command (NFEC). 1999. *Handbook for Statistical Analysis of Environmental Background Data*. San Diego: Southwest Division, Naval Facilities Engineering Command (SWDIV) and Engineering Field Activity (EFA) West. April.

HHRA REFERENCES (Continued)

Naval Facilities Engineering Command (NFEC). 2000. Navy Interim Final Policy on the Use of Background Chemical Levels. Office of the Chief of Naval Operations. September 18.

Naval Facilities Engineering Command (NFEC). 2002. Guidance for Environmental Background Analysis. Volume I: Soil. Washington, D.C. NFESC UG-2049-ENV. April.

Sander, Rolf, 1999. Compilation of Henry's Law Constants for Inorganic and Organic Species of Potential Importance in Environmental Chemistry. From Website: [Http://www.mpch-mainz.mpg.de/~sander/res/henry.html](http://www.mpch-mainz.mpg.de/~sander/res/henry.html). Air Chemistry Department. Max-Planck Institute of Chemistry. PO Box 3060. 55020 Mainz, Germany. Version 3: April 8.

Singh, Anita, A.K. Singh, M. Engelhardt, and J. Nocerino. "On the Computation of the Upper Confidence Limit of the Mean of Contaminant Data Distributions." Submitted for Publication, 1999.

United States Environmental Protection Agency (EPA). 1986a. Air Quality Criteria for Lead. Environmental Criteria and Assessment Office. Research Triangle Park, NC. EPA 600/8-83/028.

United States Environmental Protection Agency (EPA). 1986b. Guidelines for the Health Risk Assessment of Chemical Mixtures. Federal Register, Vol. 51, No. 185, p. 34014 et seq. September 24.

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

United States Environmental Protection Agency (EPA). 1989b. Guidance Document on the Statistical Analysis of Ground-Water Monitoring Data at RCRA Facilities. Interim Final Guidance. EPA 530/SW-89-026. Office of Solid Waste. Washington, DC.

United States Environmental Protection Agency (EPA). 1991a. Chemical Concentration Data Near the Detection Limit. Region III Technical Guidance Manual, EPA-3/HWMD/8-91/002.

United States Environmental Protection Agency (EPA). 1991b. Exposure Point Concentrations in Groundwater, EPA Region III Technical Guidance Manual, EPA/903/8-91/002.

HHRA REFERENCES (Continued)

United States Environmental Protection Agency (EPA). 1992a. Statistical Analysis of Ground-Water Monitoring Data at RCRA Facilities; Addendum to Interim Final Guidance. Office of Solid Waste Permits and State Programs Division. Washington, DC. July.

United States Environmental Protection Agency (EPA). 1992b. Methods for Evaluating the Attainment of Cleanup Standards. Volume 3: Reference-Based Standards for Soils and Solid Media. Office of Policy, Planning, and Evaluation. Washington, DC.

United States Environmental Protection Agency (EPA). 1992c. Guidelines for Exposure Assessment. Exposure Assessment Group. Office of Health and Environmental Assessment. In Federal Register, Vol. 57, No. 104, p. 22888. Friday, May 29.

United States Environmental Protection Agency (EPA). 1992d. Guidance on Risk Characterization for Risk Managers and Risk Assessors. Memorandum from F. Henry Habicht on February 28, 1992. Office of the Administrator. Washington, D. Publication 9285.7-081C.

United States Environmental Protection Agency (EPA). 1992e. Supplemental Guidance to RAGS: Calculating the Concentration Term. Office of Solid Waste and Emergency Response. Washington D.C. May.

United States Environmental Protection Agency (EPA). 1992f. Dermal Exposure Assessment: Principles and Applications. Interim Report. EPA 600/8-91/011/B Office of Health and Environmental Assessment. Washington D.C. January.

United States Environmental Protection Agency (EPA). 1993a. Superfund's Standard Default Exposure Factors for Central Tendency and Reasonable Maximum Exposure. Draft.

United States Environmental Protection Agency. 1993b. Selecting Exposure Routes and Contaminants of Concern by Risk Based Screening. Region III Technical Guidance Manual, EPA/903/R-93-001. Including Update April, 2003.

United States Environmental Protection Agency. 1993c. Provisional Guidance for Quantitative Risk Assessment of Polycyclic Aromatic Hydrocarbons. EPA/600/R-93/089. Environmental Criteria and Assessment Office. July.

HHRA REFERENCES (Continued)

United States Environmental Protection Agency (EPA). 1994a. Revised Interim Soil Lead Guidance for CERCLA Sites and RCRA Corrective Action Facilities. OSWER Directive 9355.4-12.

United States Environmental Protection Agency (EPA). 1994b. Guidance Manual for the Integrated Exposure Uptake Biokinetic Model for Lead in Children. EPA/540/R-93/081. Office of Emergency and Remedial Response. Washington, D.C. February.

United States Environmental Protection Agency (EPA). 1995. Assessing Dermal Exposure from Soil, EPA Region III Technical Guidance Manual, EPA/903-K-95-003. December.

United States Environmental Protection Agency (EPA). 1996a. Guidance for Data Quality Assessment. Pre-Publication Copy. Final. EPA QA/G-9. Quality Assurance Division. Washington, DC. February.

United States Environmental Protection Agency. 1996b. Recommendations of the Technical Workgroup for Lead for an Interim Approach to Assessing Risks Associated with Adult Exposures to Lead in Soil. Technical Review Workgroup for Lead. December.

United States Environmental Protection Agency (EPA). 1996c. Soil Screening Guidance. EPA/540/R-96/018. Office of Emergency and Remedial Response. Washington, DC. April 1996.

United States Environmental Protection Agency (EPA). 1997a. Health Effects Assessment Summary Tables (HEAST). EPA/540/R-95-036. Office of Solid Waste and Emergency Response. May.

United States Environmental Protection Agency (EPA). 1997b. Exposure Factors Handbook. Update to Exposure Factors Handbook. EPA/600/8-89/043 - May 1989. Office of Research and Development.

United States Environmental Protection Agency (EPA). 1998. Risk Assessment Guidance for Superfund: Volume 1. Human Health Evaluation Manual (Part D, Standardized Planning, Reporting, and Review of Superfund Risk Assessments). Office of Emergency and Remedial Response. January 1998.

United States Environmental Protection Agency (EPA). 2000. Exposure and Human Health Reassessment of 2,3,7,8-Tetrachlorodibenzo-p-dioxin (TCDD) and related compounds, Part II: Health Assessment of 2,3,7,8-Tetrachlorodibenzo-p-Dioxin (TCDD) and Related Compounds. EPA/600/P-00/001Be. Draft Final. September.

HHRA REFERENCES (Continued)

United States Environmental Protection Agency (EPA). 2001a. Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment. Interim). September. Office of Emergency and Remedial Response. Washington, DC.

United States Environmental Protection Agency (EPA). 2001b. Fact Sheet. Correcting the Henry's Law Constant for Soil Temperature, from website: <http://www.epa.gov/superfund/programs/risk/airmodel/factsheet.pdf>

United States Environmental Protection Agency (EPA). 2002a. Calculating Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites. Office of Emergency and Remedial Response. OSWER Directive 9285.6-10. December.

United States Environmental Protection Agency (EPA). 2002b. ProUCL – Version 2.1. ProUCL User's Guide. Prepared for EPA by Lockheed Martin. April.

United States Environmental Protection Agency (EPA). 2002c. Guidance for Comparing Background and Chemical Concentrations in Soil for CERCLA Sites. Office of Solid Waste, Emergency and Remedial Response. EPA 540-R-01-003. September.

United States Environmental Protection Agency (EPA). 2003a. EPA Region III Risk-Based Concentration (RBC) Tables. April 2003.

United States Environmental Protection Agency (EPA). 2003b. Integrated Risk Information System (IRIS). Database.

Van den Berg et al., 1998. Toxic Equivalency Factors (TEFs) for PCBs, PCDDs, PCDFs for Humans and Wildlife. Environmental Health Perspectives, 106 (12), 775-792.

TABLES

FIGURES