Final
Remedial Investigation Report
Landfill Areas 1 and 3
Former Lee Field Naval Air Station
Green Cove Springs, Florida

Volume 1 - Text, Tables, and Figures

Prepared For:
CESAS-CT-E
U.S. Army Corps of Engineers
South Atlantic Division
Savannah, Georgia

March 2004

Shaw™ Shaw Environmental, Inc.

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US Army Corps
of Engineers
Savannah District

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Prepared for:

CESAS-CT-E
U.S. Army Corps of Engineers
South Atlantic Division
P.O. Box 889
Savannah, Georgia 31402-0889

Prepared by:

Shaw Environmental, Inc.
312 Directors Drive
Knoxville, Tennessee 37923

March 2004

Review Signature Page

Title. Final Remedial Investigation Report, Landfill Areas 1 and 3, Former Lee Field Naval Air Station, Green Cove Springs, Florida, March 2004.

Kenn a. C

Kenneth A. Cunningham, P.G.
Shaw Task Manager/Primary Author

3/11/04
Date Reviewed

Kenn a. C for

Glenn Quarles
Shaw Project Manager

3/12/04
Date Reviewed

James W. Hamill for

Ray Clark
Peer Reviewer

3/11/04
Date Reviewed

Joyce McLee for

Maureen McMyler
Project Chemist

3/12/04
Date Reviewed

Terri Robinson

Terri Robinson
Technical Editor

3/11/04
Date Reviewed

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List of Acronyms

ABS	dermal absorption factor
Accutest	Accutest Laboratories, Inc.
AF	adherence factor
ARAR	applicable or relevant and appropriate requirement
ASTM	American Society for Testing and Materials
AT	averaging time
bgs	below ground surface
BHC	betahexachlorocyclohexane
BHHRA	baseline human health risk assessment
Black & Veatch	Black & Veatch Special Projects Corporation
BSC	background screening criterion
BW	body weight
Cal-EPA	California Environmental Protection Agency
CDI	chronic daily intake
CLP	Contract Laboratory Program
cm^2	square centimeter
COC	contaminant of concern
COPC	chemical of potential concern
COPEC	chemicals of potential ecological concern
CSEM	conceptual site exposure model
CTL	cleanup target level
D	dust loading factor
D&M	Dames & Moore
^{o}F	degrees Fahrenheit
DA	dose absorbed
DCE	dichloroethene
DDD	dichlorodiphenyldichloroethane
DDE	dichlorodiphenyldichloroethene
DDT	dichlorodiphenyltrichloroethane
DOD	U.S. Department of Defense
DOE	U.S. Department of Energy
DPT	direct-push technology
EBSV	ecological benchmark screening value
ED	exposure duration

EDQL	ecological data quality objectives
EQP	equilibrium partitioning
EF	exposure frequency
EPA	U.S. Environmental Protection Agency
EPC	exposure-point concentration
ESE	Environmental Science and Engineering, Inc.
ET	exposure time
ETDC	Environmental Technology Development Center
FDEP	Florida Department of Environmental Protection
FFWCC	Florida Fish and Wildlife Conservation Commission
FGGC	Florida Groundwater Guidance Concentrations
FI	fractional term
FID	flame-ionization detector
FNAI	Florida Natural Areas Inventory
f_{oc}	fractional organic carbon
g/m^3	grams per cubic meter
GAF	gastrointestinal absorption factor
GCTL	groundwater cleanup target level
gpm	gallons per minute
HEAST	Health Effects Assessment Summary Tables
HI	hazard index
HQ	hazard quotient
HQ_{screen}	screening-level hazard quotient
HTRW	hazardous toxic and radiological waste
ID	inside diameter
IDW	investigation-derived waste
ILCR	incremental lifetime cancer risk
IRIS	Integrated Risk Information System
IT	IT Corporation
kg	kilogram
K_{ow}	octanol/water partition coefficient
K_p	permeability coefficient
L/day	liters per day
LFNAS	former Lee Field Naval Air Station

List of Acronyms *(Continued)*

μg/kg	micrograms per kilogram
μg/L	micrograms per liter
m^2	square meter
m^3	cubic meter
MDC	maximum detected concentration
MDL	method detection limit
mg/cm^2	milligrams per square centimeter
mg/day	milligrams per day
mg/kg	milligrams per kilogram
mg/L	milligrams per liter
mg/m^3	milligrams per cubic meter
MHSPE	Dutch Ministry of Housing, Spatial Planning, and Environment
mm	millimeter
MPC	maximum permissible concentration
msl	mean sea level
NAAF	Naval Auxiliary Air Facility
NAAS	Naval Auxiliary Air Station
NCEA	National Center for Environmental Assessment
NCRP	National Council on Remediation Protection and Measurement
NTU	nephelometric turbidity unit
OD	outside diameter
PAH	polynuclear aromatic hydrocarbon
PCB	polychlorinated biphenyl
PCE	tetrachloroethene
PHR	PHR Environmental Consultants, Inc.
ppm	parts per million
PQL	practical quantification limit
PVC	polyvinyl chloride
QA	quality assurance
QC	quality control
RCRA	Resource Conservation and Recovery Act
redox	oxidation-reduction potential
RfC	reference concentration
RfD	reference dose

List of Acronyms (Continued)

RI	remedial investigation
RIVM	Dutch National Institute of Public Health and the Environment
RGO	remedial goal option
RL	reporting limit
RME	reasonable maximum exposure
SA	exposed skin surface area
SAP	sampling and analysis plan
SCL	Southern Chemists Laboratories
SCTL	soil cleanup target level
SERCC	Southeast Regional Climate Center
SF	slope factor
Shaw	Shaw Environmental, Inc.
SHP	Safety and Health Plan
SJRWMD	St. Johns River Water Management District
SLERA	screening-level ecological risk assessment
SQAG	Sediment Quality Assessment Guidelines
STC	source-term concentration
SVOC	semivolatile organic compound
SWCTL	surface water cleanup target level
TAL	target analyte list
TCE	trichloroethene
TCLP	toxicity characteristic leaching procedure
T&E	threatened and endangered
TEL	toxic effect level
TOC	total organic carbon
TRPH	total recoverable petroleum hydrocarbon
UCL	upper confidence limit
UCLP	upper confidence limit of the 95th percentile
USACE	U.S. Army Corps of Engineers
USCS	Unified Soil Classification System
USDA	U.S. Department of Agriculture
USGS	United States Geological Survey
US Waste	U.S. Waste Logistics, Inc.

List of Acronyms *(Continued)*

UTL	upper tolerance limit
VOC	volatile organic compound
WTP	water treatment plant
WRS	Wilcoxon Rank Sum
YSI	Yellow Springs Instrument Company

1.0 Introduction

Through Total Environmental Restoration Contract Number DACA21-96-D-0018, Task Order Number 9, the Savannah District of the U.S. Army Corps of Engineers (USACE) retained Shaw Environmental, Inc. (Shaw), formerly IT Corporation (IT) to conduct a remedial investigation (RI) at two former waste disposal sites, Landfill Areas 1 and 3, at the former Lee Field Naval Air Station (LFNAS). This RI report summarizes the investigative activities conducted by Shaw between September 2000 and October 2002 at the above-referenced sites.

The LFNAS occupies approximately 1,560 acres in Clay County, Florida, within the city limits of Green Cove Springs. This location is approximately 30 miles south of the City of Jacksonville on the western bank of the St. Johns River. The site is located in Section 38, Township 6 South, Range 26 East, as shown on the 7.5-minute, Green Cove Springs, Florida (United States Geological Survey [USGS], 1991) topographic quadrangle map (Figure 1-1). The LFNAS facility contains extensive docking facilities on the St. Johns River, a railroad system, four 5,000-foot aircraft runways (one of which [Runway 5–23] is still active), and numerous supporting structures.

The Department of Defense (DOD) reportedly utilized three areas for the storage and/or disposal of unknown quantities of materials during the active life of the LFNAS, from 1940 to 1963. These areas have been designated as Landfill Areas 1, 2, and 3. Landfill Area 2 was previously addressed in the *Final Remedial Investigation Report, Landfill Area 2* (Shaw, 2003a). Therefore, this RI focused on Landfill Areas 1 and 3. Landfill Area 1 encompasses approximately 6.2 acres and is located near the southern edge of the LFNAS property. Landfill Area 3 is approximately 7.4 acres in size and is located near the southwestern edge of the LFNAS property. Historical maps of the LFNAS site list Landfill Areas 1 and 3 as "Sanitary Waste Disposal" areas. Figure 1-2 is a Base-wide drawing of the LFNAS that indicates all areas under investigation. Figures 1-3 and 1-4 are Base maps detailing all pertinent features of Landfill Areas 1 and 3, respectively.

1.1 Scope of Work

The scope of the RI at Landfill Areas 1 and 3 included conducting a review of historical documents such as plans, specifications, site activity summaries, and aerial photographs; preparing site-specific work plans, such as a sampling and analysis plan (SAP) and safety and health plan (SHP); mobilizing and conducting field investigative activities; and preparing this RI report.

1.2 Project Objectives

Several preliminary assessments have been conducted at the LFNAS landfills over the past several years, but the results of these were inconclusive. The primary objective of this task order is to determine the presence and nature of contamination in groundwater, surface and subsurface soils, surface water, and sediments in the vicinity of Landfill Areas 1 and 3. During the RI, surface and subsurface soil, groundwater, surface water, and sediment samples were collected and analyzed to determine if past disposal practices resulted in releases of contaminants to the environment. In addition, both disposal areas were monitored for the presence of landfill gases. Following the receipt of analytical results, the new data was evaluated in conjunction with historical data collected at Landfill Areas 1 and 3. The analytical data was then compared to data collected as part of the installation-wide background chemical data study, previously conducted by Shaw (Shaw, 2004). The background study provided a baseline for soil, groundwater, sediment and surface water quality at the LFNAS. Another objective was to determine geotechnical characteristics and permeability of soils covering and surrounding the landfills. Human health and ecological risk assessments were performed to determine whether potential contaminants might affect receptors in the area and whether or not corrective actions are warranted at these sites.

The results of this investigation are presented in this RI Report. Chapter 1.0 presents the project description and installation history and description. The field investigation methods are described in Chapter 2.0. Chapter 3.0 provides the site-specific history, geology, and hydrogeology. The analytical results from the sampling activities and the comparison to background are discussed in Chapter 4.0. The human health and ecological screening level risk assessments are provided in Chapters 5.0 and 6.0, respectively. The summary and conclusions are provided in Chapter 7.0. References used in this report are listed in Chapter 8.0. Summary tables of historical data are presented in Appendix A. Appendix B contains the test pit logs and photographs, drilling logs, and a variance report. Well construction and development logs and development photos are provided in Appendix C. Groundwater sampling purge logs are presented in Appendix D. Geotechnical data and aquifer test calculations are provided in Appendix E and F, respectively, and survey data is provided in Appendix G. Sample collection logs are presented in Appendix H, while the analytical data quality control report, data summary and chains-of-custody are provided in Appendix I. Appendix J contains investigation-derived waste documentation. Comparisons to background reports for Landfill Areas 1 and 3 are presented in Appendices K and L, respectively. The human health and ecological screening risk assessment information are presented in Appendices M and N, respectively.

1.3 Installation Description

1.3.1 Base History

The U.S. Navy initially surveyed the site of the LFNAS in May 1939. The site was graded and had been previously used as a civilian landing field. The Navy began acquiring the property in June 1940 and by November 1940 obtained title to 1,059.6 acres. Additional acreage was acquired in 1941 and 1943 until the Base occupied a total of 1,393 acres (1,116 of "hardland" and 277 of swampland) (PHR Environmental Consultants, Inc. [PHR], 2001).

Construction of the Base began in August 1940. The original facility consisted of four, 5,000-foot runways, sixteen buildings, one small arms range and two high explosives magazines, a hanger and control tower, and a gasoline storage and distribution system. The field was completed in March 1941 and dedicated as Benjamin Lee II Field. The Base was first commissioned as a Naval Auxiliary Air Facility (NAAF) to the Jacksonville Naval Air Station. Its mission was to conduct flight training for student naval aviators. In 1943, the NAAF was redesignated as a Naval Auxiliary Air Station (NAAS) and a semi-independent station.

Following the end of World War II, LFNAS was decommissioned as a NAAS and recommissioned as U.S. Naval Station, Green Cove Springs, in December 1945. Its new mission was to accommodate and mothball vessels of the inactive naval fleet. The "mothballing" process entailed maintenance and preservation of the vessels to prevent deterioration during periods of inactivity. This provided a large reserve fleet of ships that could easily be returned to active duty on short notice. To perform its new mission, LFNAS was converted from an air station to a naval station. At this time, use of the runways was transferred to NAS Jacksonville for limited training operations support. In order to provide permanent berthing facilities for the reserve fleet, the Navy acquired 111.44 acres along the St. Johns River waterfront (20.34 acres of filled land and 91.1 acres of submerged land) and began construction of eleven 1,800-foot piers in July 1946. Prior to completion of the piers, vessels were moored in the St. Johns River. Construction of the piers was completed in 1947. The Base served as a ship mothball and refurbishing facility until 1962 (PHR, 2001). As many as 600 shallow draft fleet ships, primarily landing craft and liberty ships were berthed at the piers at one time (Environmental Science and Engineering, Inc. [ESE], 1988).

The Navy began considering possible disposal of excess property at Naval Station Green Cove Springs in 1955. The Chief of Naval Operations deactivated Naval Station Green Cove Springs

effective December 31, 1959 and transferred all functions and properties to the Florida Group, Atlantic Reserve Fleet on January 1, 1960. The Florida Group was scheduled for deactivation on June 30, 1962.

The property and improvements were declared excess by the U.S. Navy and were sold to the City of Green Cove Springs in September 1963. On July 9, 1965, the City of Green Cove Springs sold the property to J. Louis Reynolds of Reynolds Aluminum and assigned several leases to the new owner. There were few tenants and the property deteriorated until the early 1980s.
In 1981, Clay County Port, Inc. acquired the industrial park property (Reynolds Industrial Park) and continued site development. The property was annexed by the City of Green Cove Springs in August 1984 and upgraded water, sewer, and electrical services were expanded to the industrial park. Businesses currently and/or formerly operating at the LFNAS include a railroad car refurbisher, a small ship-building facility, truck driver training school, a plant nursery, aircraft maintenance facility, automobile brake testing facility, fiberglass pipe manufacturing plant, an airstrip for small airplanes, and a public golf course. Future land use at Reynolds Industrial Park is expected to consist of a combination of industrial, commercial, and possibly residential property. Today, the LFNAS area occupies approximately 1,560 acres of land.

Adjacent Land Use. The City of Green Cove Springs town center lies approximately 1 mile northwest of the LFNAS. The St. Johns River borders the former Base on the north and northeast; a trailer park is also located adjacent to the site on the northeast. Commercial and industrial facilities border the LFNAS on the west, both east and west of U.S. Highway 17. Three Mile Swamp occupies the southwest corner of the LFNAS, south and west of Wildwood Road. Cattail Creek Golf Course and the Bayard Wildlife Management Area border the site on the east. The golf course was originally constructed for use by Base personnel in the middle to late 1950s. The Bayard Wildlife Management Area is a wildlife refuge owned by the St. Johns River Water Management District (SJRWMD) and consists mainly of wooded and undeveloped land. Land use for the area surrounding the facility is shown on Figure 1-5.

1.3.2 Climate

The climate in Clay County Florida is described as "moderate" by the U.S. Soil Conservation Service (U.S. Department of Agriculture [USDA], 1989). The summers are long, hot, and humid, and the winters are cool. The coolest month is January with a mean daily temperature of 53.2 degrees Fahrenheit (°F), and the warmest is July with a mean daily temperature of 81.3 °F. The average annual rainfall is 53 inches, the majority of which falls from June through September. Tropical storms can affect the area from early June through mid-November, but

hurricane-force winds are rare because of the county's inland location. However, the heavy rains associated with tropical systems can cause flooding in low-lying areas.

1.3.3 Topography and Drainage

The LFNAS lies within the Eastern Valley of the Coastal Lowland Province in Northern Florida. The Eastern Valley physiographic region consists of mostly swamps and flatwoods (nearly level, poorly drained soils characterized by open woods of pine, saw palmetto, and pineland threeawn) (USDA, 1989) and ranges from 0 to 30 feet above mean sea level (msl) within Clay County. The Eastern Valley near the site grades into a broad, shallow regional valley approximately 15 to 20 miles wide, which slowly deepens northward until the St. Johns River drops to sea level (ESE, 1988). The Eastern Valley is bordered on the east by the Atlantic Coastal Ridge, which ranges from 25 to 30 feet above msl (ESE, 1988). West of the Eastern Valley is the Duval Upland, which ranges from 30 to 100 feet above msl.

The LFNAS facility is located on a series of beach ridge deposits associated with higher sea levels during the Late Pleistocene epoch. Topographic relief is low, with elevations gently decreasing northeastward toward the river. The central portions of the facility lie 20 to 25 feet above msl, while the northern portions of the site lie less than 10 feet above msl. The average elevation of Three Mile Swamp, located south of the facility is less than 10 feet above msl.

1.3.4 Regional Geology

Soils. Eastern Clay County is characterized by poorly drained land consisting mostly of woods and swamps. The site lies within the Sapelo-Meadowbrook-Leon general soil map unit (USDA, 1989). These soils consist of nearly level, poorly drained soils that are sandy to depths of 40 to 79 inches below ground surface (bgs), and that are found in the flatwoods. Some are characterized by loamy subsoil; some have sandy subsoil underlain by loamy subsoil; and others are sandy throughout, with subsoil coated with organic matter.

The LFNAS is underlain by several specific soil units (Figure 1-6). The northern portion of the site is underlain by the Hurricane-Urban land complex soils and the Mandarin fine sand (USDA, 1989). The surface layer of Hurricane soil in Clay County is typically very dark grayish-brown fine sand approximately 7 inches thick. The subsurface layer is comprised of grayish-brown fine sand from 7 to 11 inches bgs and pale brown fine sand from 11 to 25 inches bgs. "Urban land" denotes areas covered with buildings, streets, parking lots, sidewalks, and other structures. The urban land parts of this soil map unit are generally developed on Hurricane fine sand. The

Mandarin fine sand is nearly level and somewhat poorly drained, and is found on slight rises in the flatwoods. The surface layer is black fine sand approximately 5 inches thick, and the subsurface soil consists of light gray fine sand to a depth of 28 inches bgs.

The portion of the LFNAS that includes and surrounds the runways is underlain by the Sapelo-Urban land complex soils, which consist of nearly level, poorly drained soil and areas of urban land (USDA, 1989). In Clay County, the surface layer of the Sapelo soil typically consists of black fine sand to a depth of 4 inches bgs. The subsurface layer is gray and light gray fine sand to a depth of 19 inches. The upper part of the subsoil, found from 19 to 24 inches bgs, is comprised of black fine sand. Underlying that layer is dark reddish-brown fine sand, which extends to 32 inches bgs.

Soils of the southern and southwestern portions of the facility, including the landfill sites, consist of Leon fine sand, Osier fine sand, Allanton and Rutledge mucky fine sands (depressional), Goldhead fine sand, Plummer fine sand, and Meggett fine sandy loam (USDA, 1989). All of these soil units are described as nearly level and poorly drained.

Geology. Clay County is underlain by a thick sequence of sediment associated with the Atlantic Coastal Plain Province. Figure 1-7 shows a generalized cross-section across the Clay County area. Units underlying the area range in age from Recent to Jurassic and are more than 18,600 feet thick. Because of their occurrence at great depths, little is known about the Jurassic age rocks. The oldest rocks described as of 1986 are the Taylor and Navarro units of Late Cretaceous age. Taylor rocks consist of light gray chalk or argillaceous chalk with thin layers of dolomite. Navarro rocks consist of tan to white, soft, friable, limestone. These units are 200 to 1,300 feet thick, respectively. The top of the Navarro occurs at depths of approximately 2,000 to 2,500 feet bgs in Clay County (Miller, 1986).

The Cretaceous rocks are overlain by Tertiary Paleocene age carbonate-evaporite rocks. The primary unit of the Paleocene rocks is the Cedar Keys Formation. The lower two-thirds of the Cedar Keys consists of fine to microcrystalline dolomite with interbedded anhydrite. This lower unit comprises the lower confining unit of the Floridan Aquifer System. The upper part of the formation consists of coarsely crystalline dolomite. The Paleocene rocks are approximately 500 feet thick in the Clay County area (Miller, 1986).

Three rock units comprise Eocene age rocks in the vicinity of Clay County. The Oldsmar Formation of Early Eocene age overlies the Cedar Keys Formation. The Oldsmar is composed

mostly of micritric to finely granular limestone with interbedded fine to medium crystalline dolomite. The Oldsmar Formation is overlain by the middle Eocene Avon Park Formation. This formation is a soft to well indurated, pelletal to locally micritic limestone. The limestone is thin to thickly interbedded, fine to medium crystalline, slightly vuggy dolomite. The Avon Park Formation is exposed at the surface in Citrus and Levy Counties, representing the oldest rock outcropping in Florida. The Avon Park Formation is overlain by the late Eocene Ocala Limestone. This unit consists of an upper soft, somewhat friable, porous coquina, and a lower soft to semi-indurated, fossiliferous limestone. The Ocala is one of the most permeable units within the Floridan Aquifer as a result of karst dissolution of the rock. The total thickness of the Eocene age rocks underlying Clay County is approximately 800 feet (Miller, 1986).

A thin unit of Oligocene age rock, the Suwannee Limestone, is shown in cross-section by Miller, (1986) to be present in Clay County. The Suwannee Limestone is described as consisting of two rock types: (1) crystalline, vuggy limestone and (2) finely pelletal limestone in a micritic to finely crystalline limestone matrix. This unit is 0 to 100 feet thick and forms the uppermost rock unit of the Floridan Aquifer.

The Miocene age Hawthorn Group overlies the Eocene and Oligocene rocks in the Clay County area. The Hawthorn in Clay County is composed of three formations: (1) the basal Penny Farms Formation, (2) the middle Marks Head Formation and, (3) the upper Coosawhatchie Formation. The group ranges from approximately 75 to 90 feet thick and forms the upper confining unit of the Floridan Aquifer System (Scott, 1988). A generalized cross-section for the Hawthorn Group is detailed on Figure 1-8.

The Penny Farms Formation consists of a lower carbonate member and an upper siliciclastic member. The carbonate member is composed of variably sandy, phosphatic, clayey dolostones with interbedded sand and clay. The upper member of the Penny Farms Formation consists of olive gray, fine to coarse grained, variably phosphatic, dolomitic, silty, clayey sands and sandy clays with interbedded carbonates. The phosphate content can reach the point of being classified as a phosphorite sand (50 percent or greater phosphate grains).

The Marks Head Formation is the most complex in the Hawthorn Group. The Florida portion of the unit consists of interbedded sand, clay, and dolostone. The sands are light to dark gray, and generally fine to medium grained, dolomitic, silty, clayey, and phosphatic. The clays are greenish to olive gray, sandy, silty, dolomitic and phosphatic. The clay beds can comprise a

large portion of the formation. The dolostone portions of the formation are yellowish to olive gray, micro to finely crystalline, sandy, phosphatic, and clayey.

The Coosawhatchie Formation is similar in lithology to the Marks Head, consisting of quartz sands, clays and dolostones. The sands are phosphatic with percentages of phosphate ranging from a trace to more than 20 percent. The Coosawhatchie sediments range from light gray to greenish and olive gray in color (Scott, 1988).

Overlying the Hawthorn Group is the late Tertiary age Cypresshead Formation. The Cypresshead Formation consists of fine to coarse quartz sand. The sand is moderate to well sorted and contains some quartz gravel. Clay is commonly present in minor amounts. Mica is common in minor percentages, especially in finer sands. The Cypresshead Formation is the oldest surficial unit in Clay County, outcropping in the central portion of the county (Scott, 1992).

Overlying the Cypresshead Formation in the western portion of Clay County is the Pleistocene age Trail Ridge sands. This beach ridge and dune deposit is comprised of fine to medium grained sands. Peat beds and pieces of wood are common. The Trail Ridge sands contain ore grade concentrations of heavy minerals (Scott, 1992).

The surficial unit in eastern Clay County consists of undifferentiated Quaternary sediments. These sediments are composed of sands, clayey sands, and clays occasionally containing limited numbers of mollusk shells. This package may contain reworked Cypresshead. Lowlands along the St. Johns River and tributaries are mapped as Holocene fluvial sediments. This alluvial deposit is composed of quartz sands, silt, clay, and marls (poorly consolidated, variably sandy, clayey, shelly carbonate sediments). Peat and other organic-rich sediments are often present. This unit is common below 10 feet above msl (Scott, 1992).

Structural features affecting the Floridan Aquifer System in the Clay County vicinity include the Peninsular Arch, faulting, and the Southeast Georgia Embayment (Figure 1-9). The site is located on the east limb of the Peninsular Arch; therefore, the sediments generally dip to the east-northeast. A graben is present under the St. Johns River valley, extending approximately from the Clay-Putnam County line northward into central Duval County. The graben terminates south of the axis of the Southeast Georgia Embayment that underlies northeast Florida and southeast Georgia. The basin is a shallow east-northeast plunging syncline. The southwestern edge of the embayment extends into northern Clay County (Miller, 1986).

The most prominent feature affecting the Hawthorn Group is the Ocala Platform. The feature is located in the northwestern portion of the peninsula, west of Clay County. The platform is not a true structural feature, but rather an area of either an anomalous deposition of carbonate sediments or differential compaction of carbonate material after deposition (Miller, 1986). The Ocala Platform may be the controlling feature in the formation and deposition of phosphorites in the Florida Miocene sediments (Scott, 1988). Two structural features affect the Hawthorn Group sediments, including the Jacksonville Basin and the St. Johns Platform. The Jacksonville Basin is located in northeast Florida and is the most prominent low in the northern peninsula. Hawthorn sediments in the deepest part of the basin exceed 500 feet thick. The basin is considered a sub-basin of the Southeast Georgia Embayment. The St. Johns Platform is a broad, low ridge on the erosional surface of the Ocala Group and plunges gently towards the Jacksonville Basin. Site-specific geology for Landfill Areas 1 and 3 is discussed in Chapter 3.0.

1.3.5 Regional Hydrogeology

Clay County is underlain by a shallow unconfined and a deeper confined aquifer. The upper unconfined aquifer is contained in the Quaternary Cypresshead and recent undifferentiated sediments. The top of the shallow aquifer was encountered at depths ranging from less then 1 foot to approximately 6 feet bgs during Base-wide field activities at the LFNAS. The upper unconfined aquifer consists of approximately 75 feet of unconsolidated, undifferentiated sand, sandy clay, and shell material. Groundwater moves in this aquifer from high to low piezometric areas (under water table conditions), which generally coincides with the ground surface. Consequently, the groundwater in the unconfined aquifer generally flows toward the St. Johns River. However, due to the shallow depth to water locally and low topographic relief, frequent variations in the groundwater flow patterns are observed.

Figures 1-10 and 1-11 show the general water table surface of the surficial aquifer across the LFNAS on February 12-13, 2001 and July 10-11, 2001, respectively. A hydrologic divide is apparently present in the west-central portion of the facility, northeast of Landfill Area 2. Groundwater flow north of the divide is to the north toward the St. Johns River. South of the divide, flow is south toward Three Mile Swamp and east toward the drainage ditch along the eastern boundary of the LFNAS. Hydraulic gradients across the facility range from 0.003 feet/foot to the north to 0.005 feet/foot to the southwest.

The clay-rich Hawthorn Group separates the surficial aquifer from the deeper, confined Floridan Aquifer. The Hawthorn functions as the upper confining layer for the Floridan Aquifer, although

the Hawthorn also has permeable zones that yield substantial quantities of water. These zones are usually in the Lower Hawthorn and are often included as part of the Floridan Aquifer System (Miller, 1986).

The Floridan Aquifer System is defined by Miller (1986) as a vertically continuous sequence of carbonates rocks of generally high permeability that are mostly of middle to late Tertiary age. The rock units within the system are hydraulically connected in varying degrees and the permeability is one to several orders of magnitude greater than the upper and lower bounding (confining) units of the system. The Floridan Aquifer is divided into an upper and lower unit by a middle, discontinuous, confining to semi-confining unit. The main part of the Floridan Aquifer consists of middle to late Eocene, highly porous and permeable carbonate rocks capable of producing large quantities of water. The Floridan Aquifer consists of the following formations from oldest to youngest: the Eocene age Lake City Limestone, Avon Park Limestone, Ocala Group Limestone, and the lower permeable zones of the Miocene age Hawthorne Group. Groundwater flow in the Floridan Aquifer in the vicinity of the LFNAS is regionally from west to east (ESE, 1988).

The LFNAS is located in an area of Clay County where wells completed in the Floridan Aquifer are under artesian conditions. Floridan Aquifer artesian conditions occur along the length of the St. Johns River and in the valley and tributaries of Black Creek (Clark, et al. 1964).

Surface Water Hydrology. The St. Johns River flows northward along the eastern boundary of Clay County. Its entire reach along the county is tidally influenced, and the river stage rises and falls with each change of tide (USDA, 1989). Tides on the St. Johns River fluctuate approximately 1 foot from low to high. During times of severe storms, such as nor'easters or hurricanes, rises in ocean level can cause a significant rise in river levels and cause reverse flow in the St. Johns River as far south as Lake St. George (SJRWMD, 2000). The mean water level of the river is at sea level.

Several drainage ditches are present at the LFNAS site; some drain into the surrounding swamps and others drain into the St. Johns River (Figure 1-1). Three Mile Swamp is located in the southern portion of the facility. The area is a mature cypress swamp with an average surface elevation of less than 9 feet above msl. The water level of the swamp during the February 12, 2001 basewide water level survey was approximately 9.6 feet above msl based on the reading at a staff gauge at Landfill Area 1. The large drainage ditch located along the eastern boundary of

the LFNAS was a pre-existing creek that was the discharge outlet for Three Mile Swamp prior to development of the Base.

1.3.6 Water Supply Survey

The City of Green Cove Springs Public Works Department provides potable water for the vicinity of the LFNAS area. However, many industrial facilities and residences in Clay County use private on-site wells for their potable water supplies. The primary source of public drinking water is the confined Floridan Aquifer, but the shallow unconfined aquifer also supplies water to some wells. Water from the shallow aquifer is primarily used for irrigation purposes, but in a few cases is also used for potable water. Both aquifers exist beneath the LFNAS.

Shaw conducted a well survey during the RI investigation to determine the presence of any public or private potable wells or non-potable wells. As part of the survey, the SJRWMD was contacted to generate a database search of all permitted wells in Section 38 of Township 6 South, Range 26 East. No wells permitted by the SJRWMD were identified within a one-half mile radius of the LFNAS site.

Shaw also contacted the City of Green Cove Springs Public Works Department for information on public potable wells. According to Mr. Rob Zammataro, of the Public Works Department, the City is currently supplied potable water from two wells (denoted as RS-1 and RS-2) located on the Reynolds Industrial Park property (LFNAS). Well RS-1 is located at the existing Water Treatment Plant (WTP) No. 2, located south of State Road 16 on Cary Avenue (Figure 1-2). This well is 6-inches in diameter and cased to 274 feet, with a total depth of 657 feet bgs. Well RS-2 is located northwest of WTP No. 2 and just south of SR 16 on the LFNAS site. This well and associated pump house is listed on the 1961 Index of Structures as Building 256. Well RS-2 is also a 6-inch diameter well, cased to 294 feet, with a total depth of 668.2 feet bgs. Both of these wells supply potable water to WTP No. 2 and the water tower located at the plant. Each well is artesian and produces between 400 and 500 gallons per minute (gpm) from the Floridan Aquifer.

Potable well RS-3 is located at former WTP No. 1, located on the corner of Reynolds Boulevard and Roland Avenue at the LFNAS. This is a 6-inch diameter well cased to 276 feet, with a total depth of 654 feet bgs. The Florida Aquifer was reportedly encountered in this well at a depth of 320 feet bgs. Well RS-3 and WTP No. 1 are currently inactive, with no water tower. This well is scheduled for refurbishing for future production as a City well, pending approval by the Florida Department of Environmental Protection (FDEP). Wells RS-1, RS-2, and RS-3 were

reportedly constructed between 1939 and 1942 by the DOD and were all completed as open hole rock wells below the cased interval.

The 1961 Index of Structures indicated the presence of two additional wells at the LFNAS, listed as Structures 349 and 470 (well and pump house). The well in structure 349 is located just east of the site at the Cattail Creek Golf Course. The owners of the golf course indicated the well is currently in use for irrigation purposes only. The well is reportedly 12-inches in diameter with an undetermined casing depth and total depth. However, based on the well diameter, it is unlikely that this well was completed into the Floridan Aquifer. It is believed that this well was installed in the 1950s by the DOD when the golf course was opened. An additional well is located just west of Belle Wood Road near its intersection with State Road 16 (Structure 470) and was previously used to supply potable water for the officer's housing during the time of active Base operation. The remnants of this well are visible, with a portion of the piping present at the ground surface. The well is inactive, and it is unknown whether it has been properly abandoned. Construction details for this well have not been found. All five wells previously discussed are located greater than 1 mile from the Landfill Area 2 area and none lie downgradient of the site. The site-wide Base map (Figure 1-2) indicates the locations of these wells.

1.4 Previous Investigations

There have been several investigations conducted in the past for Landfill Areas 1 and 3. These investigations have all been preliminary studies used to identify and quantify sources of contamination. As the previous reports do not provide scaled drawings with survey coordinates for the soil samples, only approximate locations can be ascertained. Laboratory data from the previous investigations were entered into Shaw's analytical database and printed in tabular form. Analytical tables from all previous investigations are included as Appendix A.

1.4.1 EPA Site Screening Assessment of 1985

The U.S. Environmental Protection Agency (EPA), Region IV conducted a site screening assessment at Landfill Area 1 in November 1985 (EPA, 1985). For the investigation, surface and subsurface soil, sediment, and surface water samples were collected in the vicinity of Landfill Area 1. These data were subjected to only a limited quality assurance (QA) review and were, therefore, deemed qualitative.

1.4.2 Corps of Engineers 1986 Well Installation

The Mobile District USACE installed four permanent monitoring wells (W-4-86 through W-7-86) outside the estimated boundaries of Landfill Areas 1 and 3 on November 5-6, 1986. Well

W-6-86 (now LF1W-6-86) was placed northeast of Landfill Area 1 in what was believed to be a downgradient direction, based on the location of the St. Johns River. Well W-7-86 (now LF1W-7-86) was installed southwest of Landfill Area 1 in the assumed upgradient direction.

Well W-4-86 (now LF3W-4-86) was placed near the southern edge of Landfill Area 3 in what was believed to be an upgradient direction, based on the location of the St. Johns River. Well W-5-86 (now LF3W-5-86) was installed in the north-central portion of Landfill Area 3 in an assumed downgradient direction. All wells were constructed of 2-inch diameter polyvinyl chloride (PVC) with 10 feet of 0.010-inch slot screen threaded to approximately 7 to 12.5 feet of PVC riser, which includes a 3-foot PVC stickup (USACE, 1986).

1.4.3 ESE Study of 1987

ESE of Gainesville, Florida conducted a preliminary investigation at the LFNAS in September 1987. This investigation was described as a "contamination evaluation" and involved determining the presence or absence of contaminants at Landfill Areas 1 and 3, as well as other sites at the LFNAS (ESE, 1988). The ESE investigation at Landfill Area 1 included sampling the two existing monitoring wells, the collection of two subsurface composite soil samples (LFSS6 and LFSS7) from within the estimated landfill boundaries, and the collection of three surface water samples (LFSW1, LFSW2, and LFSW3). A concentration of 47 micrograms per liter (μg/L) total lead was detected in groundwater collected from well W-6-86. At that time, the groundwater standard was 50 μg/L. The soil samples were collected from approximately 2 feet bgs to the shallow water table (approximately 3 feet bgs). Low concentrations of several metals, petroleum hydrocarbons, and bis(2-ethylhexyl)phthalate were detected in the two soil samples. Polychlorinated biphenyl (PCB) Aroclor 1260 (210 micrograms per kilogram [μg/kg]) was also detected in soil sample LFSS6. Silver was detected in surface water sample LFSW2 (25 μg/L) in excess of surface water standards.

The ESE investigation of Landfill Area 3 included sampling the two existing monitoring wells and the collection of two subsurface composite soil samples (LFSS4 and LFSS5) from within the estimated landfill boundaries. The soil samples were collected from approximately 2 feet bgs to the shallow water table (approximately 3 feet bgs). Concentrations of 46 μg/L chromium, 44 μg/L lead, and 37.6 μg/L methylene chloride were detected in groundwater collected from well W-4-86. Both chromium and lead were also detected in well W-5-86 at concentrations of 103 μg/L and 62 μg/L, respectively. At that time, the groundwater standard for both chromium and lead were 50 μg/L. Low concentrations of several metals and bis(2-ethylhexyl)phthalate were

detected in soil sample LFSS4. Due to its age, the ESE data from Landfill Areas 1 and 3 were not included in the risk assessments or for comparison to background concentrations.

1.4.4 Dames & Moore Study of 1991

Dames and Moore (D&M) conducted a preliminary contamination evaluation at the LFNAS in December 1991 (D&M, 1992). The investigation included a resampling of the existing wells at Landfill Areas 1 and 3, previously installed by the USACE. The investigation at Landfill Area 1 detected barium in both W-6-86 and W-7-86, with lead also found in well W-6-86. The concentrations detected did not exceed the Florida Groundwater Guidance Concentrations (FGGC) at that time.

The investigation at Landfill Area 3 resulted in the detection of chlorobenzene at a concentration of 21 μg/L in well W-5-86. This concentration exceeds the FGGC of 10 μg/L at that time. The QA/quality control (QC) check indicated that chloroform contamination by the laboratory may have resulted in an anomalous reading of compounds containing chloroform, such as chlorobenzene. Other compounds detected in wells W-4-86 and W-5-86 include several metals, 1,4-dichlorobenzene, and acenaphthene.

1.4.5 Southern Chemists Laboratory Report of 1993

Southern Chemists Laboratories (SCL) suggested in 1993 that the metals concentrations reported as contamination in the LFNAS samples might be naturally occurring. Four soil samples were re-analyzed after using a different digestion method that left mineral sands in the sample intact. The resulting analyses detected metals concentrations below regulatory limits. The lab concluded that the metal concentrations were the result of solutioning of the naturally occurring mineral sands, and not from a contaminant source. This was reinforced by the fact that lead and chromium concentrations detected did not exceed FGGC in 1992, but were reported as such in 1987 for some samples (SCL, 1993).

1.4.6 EPA Expanded Site Inspection of 1998

Black & Veatch Special Projects Corporation (Black & Veatch) was contracted by the EPA to conduct an expanded site inspection of Reynolds Industrial Park in 1998 (Black & Veatch, 2000). The investigation included several sites at the LFNAS, including Landfill Areas 1 and 3. Samples of groundwater (RP-TW-03), surface soil (RP-SS-02 and -03), and subsurface soil (RP-SB-02 and -03), and sediment (RP-SD-02 and -03) were collected at Landfill Area 1 and analyzed for volatile organic compounds (VOC), semivolatile organic compounds (SVOC), metals, pesticides, PCBs, dioxins, and furans. The groundwater sample was collected from a

temporary well placed on the western portion of Landfill Area 1. This location coincided with surface and subsurface soil samples SS and SB-03. Soil samples SS and SB-02 were collected from approximately the east-central portion of the landfill. Sediment samples SD-02 and SD-03 were collected from the western and eastern sides of Landfill Area 1, respectively. No compounds were detected above Federal Drinking Water Standards in the groundwater sample. Concentrations of PCBs in surface soil sample RP-SS-03 and both subsurface soil samples exceed residential soil cleanup target levels (SCTL). Sample RP-SB-02 also contains barium and lead concentrations exceeding the SCTLs. Concentrations of lead, zinc, and PCBs in sediment sample RP-SD-02 and nickel, zinc, and PCBs in sample RP-SD-03 exceed screening values.

At Landfill Area 3, one surface (RP-SS-04) and one subsurface (RP-SB-04) soil sample were collected and analyzed for the same parameters as at Landfill Area 1. No compounds were detected in the surface or subsurface soil samples that exceed SCTLs. The sample locations for the landfills are approximate as indicated on the maps, as there is no survey data available from Black & Veatch.

2.0 Remedial Investigation Methodology

An installation-wide SAP (IT, 2000a), the RI work plan for Landfill Areas 1, 2, and 3 (IT, 2000b), and a Phase II Investigation work plan (IT, 2002) were prepared in accordance with the USACE, Savannah District SAP and USACE EM200-1-3. In cases where changes to the work plan methodologies occurred, a variance report was prepared and submitted to the USACE. One variance requesting a change to the installation procedures for the deep monitoring wells was generated and approved. The contents of this variance are detailed in Section 2.2.4.2.

2.1 Landfill Boundary Delineation

Prior to the initiation of any intrusive investigative activities, the approximate boundaries of Landfill Areas 1 and 3 were defined. Delineation of the landfill boundaries is critical for the proper placement of soil borings and groundwater monitoring wells. Initially, historical aerial photographs were reviewed to identify the landfill area and its approximate boundaries over time. To verify the boundaries and check for the presence of buried debris, a series of test pits were excavated by a small backhoe, outside and inside the presumed landfill areas.

2.1.1 Aerial Photograph Review

Prior to the selection of soil boring and monitoring well locations, a series of aerial photographs were reviewed to track the development and use of the Landfill Areas 1 and 3 over time. The photographs were interpreted to identify land features and activities that were indicative of former disposal sites. Typically, former disposal areas are characterized by excavations, piles, and cleared areas.

Black and white aerial photographs covering the LFNAS area for the period 1940 through January 2001 were obtained from several sources including: the National Archives, Department of the Navy, Clay County Property Appraiser Office, and the Florida Department of Transportation. Characteristics such as vegetation patterns and areas of disturbed land were documented and compared to historic Base maps. Identification of these features generally aided the selection of specific sample locations.

In addition, stereographic aerial photograph pairs from selected dates (1958 through 1990) were used to help identify features at the landfill areas by viewing them in three dimensions. This enabled a more thorough review of the landfills from the operational period just prior to Base deactivation up to the present.

As a follow-up, site walks were conducted to document and compare the current ground conditions to historical conditions shown on the aerial photographs. The results of the aerial photograph review are discussed in Sections 3.1.2 and 3.2.2.

2.1.2 Test Pit Excavation

The aerial photograph review gave a good indication of the boundaries of Landfill Areas 1 and 3. In order to confirm the western perimeter of Landfill Area 1 and the boundaries of Landfill Area 3, as indicated by the aerial photographs, a series of test pits were excavated. Due to the unknown extent of possible debris burial and content, test pits were initiated outside the assumed landfill boundary first and then continued inward. Test pits were excavated inside the boundaries to ascertain the types of material that were disposed of in the landfills.

A total of 4 test pits were excavated at the western end of Landfill Area 1 and 22 test pits were dug in and around Landfill Area 3. Test pitting was conducted using a Bobcat-mounted backhoe to depths ranging from 3 to 7 feet bgs. The test pit locations are shown on Figures 1-3 and 1-4. In most cases, the water table was encountered at depths of approximately 0.5 to 2 feet bgs. Test pit logs were used to record: the test pit locations, soil/debris types encountered, depth to water, and vapor monitoring results (Appendix B). Each test pit was also photographed to document the excavations and any debris observed. Upon completion, all test pits were backfilled to grade with the removed soil and compacted with the backhoe bucket. Test pit locations were then flagged for location by survey. Test pit findings are discussed in Sections 3.1.3 and 3.2.3.

2.2 Sampling and Field Procedures

The procedures for field activities are discussed in the following sections. Chapter 5.0 of the SAP (IT, 2000a) provides detailed instructions for sample collection and data acquisition activities that were followed during the Landfill Areas 1 and 3 RI.

2.2.1 Soil Quality Sampling

Surface and subsurface soil samples were collected to assess the presence or absence of contaminants in the vicinity of Landfill Areas 1 and 3 and to evaluate the potential risk to humans and the environment posed by media that they are commonly exposed to during routine activities. During the RI, surface soil samples were collected from within the boundaries of Landfill Areas 1 and 3 and sent for laboratory analysis. Surface soil samples were collected from the uppermost 12 inches of soil only (0 to 1 foot bgs). In addition, eight soil borings were drilled along the boundaries of Landfill Areas 1 and 3 (Figures 1-3 and 1-4).

Surface and subsurface soil samples were collected at the soil borings. Subsurface soil samples were collected from depths ranging from 1 to 3 feet bgs, or just above the static water table. All soil borings were drilled with a decontaminated stainless-steel hand auger. Sufficient soil volumes from surface and subsurface samples were retrieved for both headspace analysis by flame-ionization detector (FID) and fixed-base laboratory analysis. Each soil sample was split and placed in two clean 1-pint glass jars for measurement of headspace readings. Each jar was covered with foil and allowed to equilibrate for no less than 5 minutes prior to the FID probe insertion. One FID probe was inserted into each jar: one that detected all organic vapors and one that filtered out hydrocarbon vapors with granular activated carbon. A net FID reading was obtained by subtracting the filtered reading from the unfiltered.

Because of the high water table at the site (generally 4 feet bgs or less), only one subsurface soil sample from each soil boring was submitted for laboratory analysis. Surface and subsurface soil samples were analyzed for the following parameters: VOCs, SVOCs, pesticides, PCBs, total recoverable petroleum hydrocarbons (TRPH), target analyte list (TAL) metals, and 20 percent of the samples for polynuclear aromatic hydrocarbons (PAH). In addition, ten of the twenty surface soil samples from each site were also analyzed for total organic carbon (TOC). Samples collected for volatile organic analysis were transferred directly to Encore® samplers as soon as the sampling device was opened, then immediately placed on ice. All remaining soil for analysis was homogenized according to the procedure described in Section 6.1.1 of the SAP. These samples were placed in the appropriate laboratory-supplied sample containers, labeled for proper identification, and packed in an iced cooler for transport.

The soil was visually logged in accordance with American Society for Testing and Materials (ASTM) D-1452-85 using the Unified Soil Classification System (USCS) and as described in Section 6.1.1 of the SAP (IT, 2000a). Field observations (e.g., presence of staining, debris on the surface, etc.) were recorded in the field notes and on the soil sample collection logs (Appendix H). Hazardous, toxic, and radiological waste (HTRW) drill logs were not completed for the soil borings because they were typically less than 4 feet deep.

Selected soil samples from the vicinity of Landfill Areas 1 and 3 were analyzed for geotechnical properties. A grain size analysis was conducted for five of the surface soil samples collected within the landfill areas and five surface soil samples collected from the soil borings at each site. All geotechnical samples were sent to Shaw's Environmental Technology Development Center (ETDC) in Oak Ridge, Tennessee for analysis. Results of the geotechnical analyses are

discussed in Section 3.1.5 and 3.2.5. A copy of the geotechnical data report is included in Appendix E.

2.2.2 Lithologic Soil Sampling

Six deep borings were completed by direct-push technology (DPT) near the perimeters of Landfill Areas 1 and 3 during the initial investigation in 2000 through 2001. The DPT borings were placed on all sides of the landfills to provide adequate coverage. The borings (LF1PZ04 and LF1PZ05) along the south side of Landfill Area 1 were placed on the low-lying portion of the landfill because of the soft conditions in Three Mile Swamp that borders the site. In addition, eight DPT borings were installed in the southeastern portion of Landfill Area 3 during the Phase II investigation in 2002. The purpose of the soil borings was to determine the subsurface lithology in the vicinity of Landfill Areas 1 and 3, which would be critical for the proposed direct-push groundwater sampling and subsequent well installation activities.

Soil samples for lithologic purposes were collected with a macro-core sampler, which retrieves soil cores measuring 1.5 inches in diameter by 48 inches long. For easier collection and description, the soil cores were collected in clear plastic inserts placed inside the macro-core samplers. Descriptions were recorded on the HTRW drilling logs provided in Appendix B. Excess cuttings generated during the drilling of soil borings were drummed for later disposal.

2.2.3 Direct-Push Groundwater Sampling

Groundwater samples were collected for screening purposes using DPT at Landfill Areas 1 and 3. The groundwater samples were collected from approximate depths of 20 and 40 feet bgs, adjacent to the lithologic boring locations described in Section 2.2.2. The exact sample interval selected at each location was determined by the lithologic borings that preceded the groundwater sampling. The purpose of the direct-push groundwater sample collection was to determine the presence of any potential contaminants at the perimeters of the landfills and, if present, delineate the vertical extent. The results of the direct-push sampling were also used to help determine the location and depth of the permanent shallow and deep monitoring wells.

The groundwater samples were obtained by pushing a 4-foot stainless-steel screen encased in a 1.75-inch diameter casing. Upon reaching the desired depth, the rods and casing were retracted to expose the screen. Polyethylene tubing was inserted through the rods into the screened interval and connected to a peristaltic pump. The groundwater was pumped until clear, then the samples were collected. The Phase I samples were analyzed for VOCs and SVOCs, with 20 percent of the samples also analyzed for PAHs. Direct-push groundwater samples collected at Landfill Area 3 during the Phase II investigation were analyzed for VOCs only. The SVOC and

PAH samples were collected directly through the sampling pump and tubing. The VOC samples were collected by the pipette method of filling the tubing with the peristaltic pump and allowing gravity to drain the sample tubing directly into the sample bottles. All sample jars were labeled, placed in zip-lock bag, stored in a cooler packed with ice, delivered to the field office, and prepared for shipment to the laboratory. Sample collection logs are included in Appendix H.

2.2.4 Well Installation

Following the lithologic soil determination and direct-push groundwater sampling, six piezometers were installed (Figures 1-3 and 1-4) around each site. The piezometers were placed directly adjacent to the 2000 through 2001 direct-push groundwater sample locations on all sides of the landfills and outside the boundaries, where possible. The piezometers were used with the two existing monitoring wells at each site to determine the direction of groundwater flow, prior to permanent well installation. In conjunction with the direct-push groundwater sampling, the piezometers enabled Shaw to select the appropriate depth and placement of the permanent monitoring wells. The installation of the piezometers and monitoring wells included:

- Drilling boreholes using direct-push, hollow-stem auger, and mud rotary drilling methods.
- Containerizing cuttings and any other fluids, such as decontamination water, into steel drums.
- Installing screen, riser, surface casing, and well protection.

All piezometers were installed using DPT. The following sections provide additional information regarding installation, construction, development, and sampling of the piezometers/monitoring wells. All activities associated with installation and sampling of piezometers and monitoring wells followed Section 5.2 of the SAP (IT, 2000a).

2.2.4.1 Piezometer/Monitoring Well Installation

As discussed previously, a total of six shallow piezometers per site were installed near the perimeters of Landfill Areas 1 and 3 for the RI. Based on the groundwater analytical, lithologic, and potentiometric data obtained during the initial stages of the RI, it was determined that some of the piezometers could be used as permanent groundwater monitoring points. Because of their location relative to the groundwater flow direction, three piezometers were converted to permanent monitoring wells as follows: LF1PZ02 to LF1MW02, LF3PZ01 to LF3MW01, and LF3PZ05 to LF3MW02 (Figures 1-3 and 1-4). The remaining piezometers were left in place

with no surface protection, except a bentonite or fine sand seal and can be used for measuring water levels or later converted to a permanent well, if necessary.

Based on groundwater elevation and direct-push groundwater analytical data, one deep and four permanent shallow monitoring wells were installed at Landfill Area 1, and one deep and three shallow monitoring wells were installed at Landfill Area 3 in 2000 through 2001. After further examination of the initial RI data and the results of the 2002 DPT groundwater sampling, an additional shallow (LF3MW06) and a new deep monitoring well (LF3DW02) were installed at Landfill Area 3 in 2002. The following section describes the installation of the shallow piezometers, their conversion to permanent monitoring wells, and installation of shallow and deep monitoring wells.

Borehole Drilling. All piezometer/monitoring well locations were sampled continuously for lithologic description prior to the direct-push groundwater sampling as discussed in Section 2.2.2. As discussed in Section 2.2.2, lithologic soil samples were collected at each piezometer/well location. Descriptions were recorded on the HTRW drilling logs provided in Appendix B. Excess drill cuttings generated during the installation of soil borings, piezometers, and monitoring wells were drummed for later disposal.

The boreholes for the piezometers/wells were completed by driving 3.25-inch diameter steel casing to the required depth with a direct-push rig. The well material was assembled and lowered into the casing. An expendable, stainless steel drive point was pushed off the end of the casing and then the casing was extracted. Shallow piezometers/monitoring wells were installed to screen above the top of the water table.

An undisturbed soil sample was collected from the shallow saturated zone at one piezometer/monitoring well location at each site for geotechnical analyses. The sample was collected by DPT using a macro-core soil sampler as described previously, from depths ranging from approximately 6 to 10 feet bgs. The sample was collected in a clear plastic sleeve insert placed inside the macro-core sampler. The ends of the insert were sealed to prevent the loss of moisture and disturbance to the sample. The saturated zone geotechnical sample was analyzed for grain size, moisture, bulk density, porosity, and vertical conductivity. All geotechnical samples were sent to the ETDC for analysis. A copy of the geotechnical report is included in Appendix E.

Piezometer/Well Construction. All piezometers/wells were constructed using flush-jointed PVC screen and casing, conforming to the requirements of ASTM D 1785 for Schedule 40 pipe. Use of PVC as well materials will not compromise the objectives of the RI since PVC is non-reactive with groundwater high in sulfide, and will not be impacted by petroleum hydrocarbon product or dissolved-phase aromatic hydrocarbons at the concentrations present in groundwater (Driscoll, 1986).

Shallow piezometers/monitoring wells were constructed using 10 feet of 1-inch inside diameter (ID) by 2.5-inch outside diameter (OD) prepacked PVC screen. The screen size was 0.010-inch factory slotted screen with Schedule 40 flush-threaded casing. Filter material for the prepacked screens was 20 to 30- mesh, clean silica filter sand graded so that less than 5 percent were finer than 0.010-inch diameter. The filter pack was installed inside the 2.5-inch OD, Schedule 40 PVC, 0.010-inch slotted screen. A 1- to 3-foot thick fine sand seal was placed above the top of the prepacked screen to seal the borehole. The remaining borehole annulus was grouted by gravity method to one foot below surface with a Portland cement mix. Construction diagrams were completed for each piezometer/monitoring well installed and is provided in Appendix C. Table 2-1 summarizes the well construction of all the newly installed monitoring wells. All well installation activities were conducted in accordance with USACE specifications (USACE, 1998).

2.2.4.2 Deep Monitoring Well Installation

One deep monitoring well was installed at Landfill Area 1 and two deep wells were installed at Landfill Area 3 to serve as vertical delineation for any contaminants detected in the shallow wells. The surface casings installed in 2000 through 2001 were installed using mud rotary drilling methods because of concerns over flowing sands encountered during direct-push operations. Since installation of double-cased wells by mud rotary methods was not proposed in the work plan, a variance report (Appendix B) was prepared and approved by the USACE and FDEP prior to deep well installation. A Mobile B-57 truck-mounted drill rig was used for the mud rotary drilling.

Little contamination was detected in the groundwater during the direct-push sampling at Landfill Area 1, and there was a delay in analytical reports from the laboratory for the Landfill Area 3 direct-push sampling. Therefore, the Phase I deep wells were placed on the downgradient sides of Landfill Areas 1 and 3, adjacent to wells LF1MW04 and LF3MW03, respectively. Well LF3DW02 installed during the Phase II investigation was installed on the east side of the site, based on the analytical results from the Phase II direct-push groundwater sampling. Based on the lithology observed during the soil borings, the deep wells were screened from 40 to 45 feet

bgs. Since no clay layer was penetrated during the drilling of these wells, the 4-inch diameter surface casing was set below the screened interval of paired shallow wells.

Borehole Drilling. Soil samples were collected for lithologic determination prior to the installation of the deep well during the direct-push soil and groundwater sampling. Soil samples were collected using a split-barrel sampler, in compliance with ASTM D 1586-84. Descriptions of soils followed Section 6.1.1 of the SAP and were recorded on the HTRW drilling log provided in Appendix B. Initially, the borehole for the deep well was drilled using mud rotary methods with an 8-inch diameter tri-cone roller bit. Based on the known lithology in this location, a 4-inch diameter surface casing was installed in the borehole at depths of 20 to 25 feet bgs. The casing was grouted in place and allowed the set for 24 hours. After the grout had cured, the borehole was advanced below the casing using DPT and the wells were installed to depths of approximately 40 to 45 feet bgs.

Undisturbed soil samples were collected from the shallow and deep saturated zones of the deep monitoring well for geotechnical analyses. The samples were collected by macro-core soil sampler as described previously, from the shallow saturated zone (6 to 10 feet bgs) and the deep saturated zone corresponding to the screened intervals (40 to 45 feet bgs). The saturated zone geotechnical samples were analyzed for grain size, moisture, bulk density, porosity, and vertical conductivity. All geotechnical samples were sent to the ETDC for analysis. A copy of the geotechnical report is included in Appendix E.

Well Construction. The deep wells were double-cased using a 4-inch ID, PVC surface casing. The outer casings were set at depths of 20 to 25 feet bgs. The surface casings were grouted in place and allowed to cure for a minimum of 24 hours before the boreholes were advanced below the casings for installation of the wells. The deep wells were constructed using a 5-foot section of 1-inch ID x 2.5-inch OD prepacked PVC, 0.010-inch slotted screen threaded to 1-inch ID, Schedule 40 PVC casing. Two to three feet of fine sand was installed above the top of the prepacked screen. A second seal consisting of 2 feet of hydrated Wyoming bentonite chips was installed to seal the lower water-bearing zone where the well penetrated a clay unit. The total depth of the deep well was determined by the results of the direct-push lithologic sampling. The deep wells extend to total depths of approximately 40 to 45 feet bgs.

The remainder of the borehole above the bentonite seal was grouted using a bentonite-cement grout conforming to the following specifications:

- Portland cement conforming to ASTM C 150
- No more than 7 gallons of water per bag of cement
- Two percent by weight (2 pounds) bentonite powder.

Grout was pumped by tremie pipe into the borehole until undiluted grout flowed freely from the annular space. Once grouted, the well was developed in accordance with procedures outlined in the following section. Well construction logs are provided in Appendix C. Table 2-1 summarizes the well construction details of all the newly installed monitoring wells.

2.2.4.3 Surface Completions

The below ground installation for the new piezometers and monitoring wells at Landfill Areas 1 and 3 are exactly the same. A conversion from piezometer to permanent monitoring well only required a permanent surface completion. As the piezometers/wells installed for the RI at Landfill Areas 1 and 3 are all located in areas covered by vegetation, all piezometers/wells were completed above ground for visibility. During installation, enough riser pipe was attached to each piezometer/well screen for approximately a 3-foot length of PVC stickup. For all wells a 5-foot length of 4-inch ID protective steel casing with a locking lid was installed over the well pipe to a depth of approximately 2 feet bgs. Sand was placed within the steel protective casing and outside the well casing, to a height of 0.5 foot above the external concrete pad. A 3-foot square by 6-inch thick concrete pad was constructed around and sloping away from the well. An internal drainage hole was drilled through the steel casing just above the sand. After the grout set, the protective steel casings were painted yellow and identified by number with an aluminum well tag attached inside the protect cover lid. Three concrete-filled steel posts were set radially around the concrete pad to protect the well from damage. The steel protective casing for each well was secured with a brass lock to prevent tampering.

2.2.5 Monitoring Well Development

After the new wells were installed and grouted in place for at least 48 hours and no more than 7 days, the monitoring wells were developed. In addition, since the condition of the pre-existing wells was unknown, they were also re-developed to remove any sediment. The procedure described for well development is in accordance with Chapter 6.0 of USACE Manual No. 1110-1-4000, dated November 1, 1998 (USACE, 1998). A detailed discussion of well development is provided in Section 5.2 of the SAP (IT, 2000a).

Development was completed by pumping with a peristaltic pump and surging. Development was considered complete when the water ran clear to the unaided eye; the thickness of sediment in the well was less than 5 percent of the length of the well screen; field parameters, such as

temperature, pH, turbidity, and conductivity stabilized; and a minimum of 5 well volumes had been removed from the well. A Horiba™ U-10 water quality meter and an Orion Model 290A conductivity/Eh meter were used for well development during the Phase I investigation to determine individual water quality readings. A Yellow Springs Instrument Company (YSI) Model 6920 water quality meter with a flow-through cell was used to measure field parameters during the Phase II investigation. If all development criteria were not achieved after two attempts, the USACE was notified for further direction. Well development water was drummed until analytical results indicated the proper method for disposal, as described in Section 2.5.

Descriptions of the well development technique and physical characteristics such as clarity, color, turbidity, and odor of the water were recorded on well development logs. Water quality parameters including temperature, pH, conductivity, dissolved oxygen, oxidation-reduction potential (redox) and turbidity were recorded once per well volume until five well volumes were reached. After five well volumes, parameters were recorded every 10 minutes until development was complete. Following well development, a clear glass jar of the final development water was collected from each well and labeled. Each jar was agitated to show any suspended material in the water and then photographed. The well development logs and photographs of the final development water are presented in Appendix C.

2.2.6 Monitoring Well Groundwater Sampling

Groundwater samples were collected from all pre-existing and newly installed monitoring wells to evaluate current water quality and verify the previous groundwater analytical results. A well development/purge log and groundwater sample collection log was completed for each sample. All wells were purged and sampled as specified in Appendix D and Section 6.1.3 of the SAP (IT, 2000a). A Horiba U-10 meter and an Orion conductivity/redox meter were used for purging during the Phase I investigation to determine the sample temperature, pH, specific conductivity, redox, dissolved oxygen, turbidity, and salinity. A YSI Model 6920 water quality meter with a flow-through cell was used to measure field parameters during the Phase II investigation. Two samples were collected for each metals analysis, filtered and unfiltered. As stated in the work plan, the intent was to analyze the unfiltered sample first and only run the filtered aliquot if total concentrations exceeded state standards. However, a decision was made during the investigation to analyze the unfiltered and filtered aliquots of each groundwater sample.

Wells were purged and sampled by low-flow methods using a peristaltic pump and Teflon™-lined tubing. Purging continued until field parameters stabilized as specified in the SAP (IT, 2000a) and turbidity was reduced below 10 nephelometric turbidity units (NTU). Well purging

logs are presented in Appendix D. After parameters had stabilized, groundwater samples were collected by discharging directly into the appropriate containers for all analytes except VOCs. Samples for VOCs were collected using the pipette method as described previously in Section 2.2.3. All groundwater samples collected from the permanent wells were submitted for laboratory analysis for: VOCs, SVOCs, 20 percent for PAHs, TAL metals (total and dissolved), TRPH, pesticides, and PCBs. Wells LF1MW04 and LF1DW01 were resampled during the Phase II investigation and analyzed for VOCs only in accordance with the Phase II investigation work plan (IT, 2002). The sample containers were labeled, sealed in zip-lock bags and immediately placed in an ice-cooled cooler. Sample collection logs are included in Appendix H.

2.2.7 Surface Water/Sediment Sampling

Four surface water and sediment samples were collected from the vicinity of Landfill Areas 1 and 3. The sediment and surface water sample locations were collocated. The Landfill Area 1 surface water and sediments samples were collected from the main ditch along Wildwood Road (LF1SW/SD01 and LF1SW/SD02), the creek extending along the eastern boundary of the site (LF1SW/SD03), and from Three Mile Swamp along the southern boundary (LF1SW/SD04). At Landfill Area 3, the four sediment and surface water samples were collected from two small ponds within the landfill area and along a small ditch, which drains the pond towards the west.

Surface water samples were collected by either direct discharge into the sample containers or using a decontaminated PVC dipper. Sediment samples were collected using either a decontaminated stainless-steel spoon or hand auger depending on the depth of the surface water and placed in stainless-steel mixing bowls. The sediment was then mixed and placed in the appropriate containers. Surface water samples were collected before sediment samples. Samples requiring VOC analysis were collected in Encore samplers. Sediment samples were analyzed for VOCs, SVOCs, (all samples by EPA 8270C and 20 percent of those by EPA 8310 also), TAL metals, TRPH, pesticides, PCBs, and TOC. Surface water samples were analyzed for VOCs, SVOCs (all samples by EPA 8270C and 20 percent of those by EPA 8310), TAL metals (total and dissolved), TRPH, pesticides, PCBs, hardness, and alkalinity.

2.2.8 QA/QC Sampling

To ensure the reliability of field sampling procedures and materials, field QA/QC samples were collected or prepared as appropriate for each medium sampled, each sample shipment, and/or each sampling event. The field activities were considered one event. Section 6.1.6 of the SAP provides specific information regarding QA/QC samples and describes how to identify and document them.

2.2.9 Sample Custody, Shipping, and Tracking Procedures

All sample custody and tracking procedures, including laboratory notification, field custody procedures, identification, and shipping were performed as specified in Section 6.2 of the SAP (IT, 2000a). Most samples were pre-assigned a unique sample number prior to the start of fieldwork. Upon collection, each sample container was labeled, sealed in a zip-lock bag, placed in a cooler packed with ice, and recorded on a field activity daily log. A sample collection log was completed. The sample interval was also recorded on the HTRW drill log, if appropriate. The samples were entered into Shaw's Environmental Data Management System by the sample coordinator and a chain-of-custody form was generated. Samples were packed in coolers chilled to 4 degrees Celsius, sealed with signed and dated custody seals on the outside of the coolers, and shipped by either laboratory courier or express overnight delivery to the laboratory. The laboratory and the Shaw data manager were notified of sample shipments the same day coolers were sent.

2.2.10 Groundwater Elevation Measurements

Water level measurements were recorded for the piezometers/wells at Landfill Areas 1 and 3 on several occasions during the RI. The depths to water were measured to the nearest 0.01-foot with a decontaminated Heron water level indicator. Initially, water levels were recorded on November 8 and 27, 2000 following the installation and development of the piezometers. These measurements enabled an accurate direction of groundwater flow to be ascertained for the placement of permanent wells. The direction of groundwater flow indicated that the existing piezometers were placed such that they could be used as permanent monitoring points. Following the conversion of the piezometers to permanent wells, addition of the deep well, and development of the new wells, water levels were measured in all wells present at Landfill Areas 1 and 3. In 2001, water levels were measured on January 14, February 12, July 11, and October 23. Water levels were also measured at Landfill Areas 1 and 3 wells on October 8, 2002. The measurements were not recorded until the wells had stabilized after development, or for at least 24 hours. The water level measurements were used to confirm the flow direction and track seasonal fluctuations of the water table over time.

In addition to site-wide water levels, Base-wide events were conducted on February 12-13, July 10-11, and October 22-23, 2001, and October 6-8, 2002. These events were used to determine a Base-wide configuration of the top of the water table aquifer.

2.2.11 Aquifer Testing

Aquifer testing conducted during the RI included a Base-wide tidal influence study, slug testing, and 100-minute pump testing. The tidal influence study was conducted over an approximately 4½-day period from January 23 to January 28, 2001. The Base-wide study was performed using In Situ miniTROLL data loggers. All of the data loggers were preset prior to installation to begin recording data at the same time. One miniTROLL was deployed in the St. Johns River through the rusted hull of an abandoned barge near the Fire Training Area, approximately 1 foot above the river bottom. This position protected the device from rough wave action. In addition, one miniTROLL was installed in a selected well at each HTRW site, Underground Storage Tank Area 2, and Landfill Area 2. The instruments were positioned about 1 to 2 feet above the bottom of the well. The nearest monitoring point to Landfill Area 1 was well PAMW04 at the Pyridine/Burn Area. The tidal study area did not extend south of Wildwood Road. A laptop computer was used to check the status of the data logger periodically during the study and to terminate logging and download the data before the instruments were removed from the wells and river. Results of the basewide tidal study are discussed in the Final Landfill Area 2 RI Report (Shaw, 2003a) and Draft Multi-HTRW Site Investigation Report (Shaw, 2003b).

Slug tests provide an estimate of hydraulic properties of an aquifer in the immediate vicinity of a well where the test is conducted. The slug tests involve causing an instantaneous change in the well's static water level and measuring the recovery of the water level with time (Freeze and Cherry, 1979). The slug tests at Landfill Areas 1 and 3 were conducted on January 26, 2001 and followed the procedures described in Section 5.4 of the SAP (IT, 2000a).

The slug tests were performed using both previously installed 2-inch diameter shallow wells at Landfill Area 1 (LF1W-6-86 and LF1W-7-86) and Landfill Area 3 (LF3W-4-86 and LF3W-5-86). The following equipment was used to perform the tests:

- In Situ Hermit 3000 data logger with a pressure transducer
- A solid slug constructed of 1-inch diameter PVC pipe filled with potable water, capped and sealed
- Nylon rope
- Heron water level indicator.

The slug tests were conducted by first measuring the static water level from the top of casing. The pressure transducer was then installed and referenced. Next, the slug was installed and the

water level in the well allowed to return to within 0.01 foot of static level. After the water level had stabilized, the reference was rechecked, the data logger initiated, and the slug quickly removed. At least two tests were performed on each well for QC purposes.

After the tests were completed, the data was downloaded from the data logger to a diskette. The data was graphed and inspected for consistency and completeness. Slug test data was evaluated by the Bouwer and Rice Method (1976) using Aqtesolv for Windows Pro version 3.0.

The deeper aquifer zone was scheduled for testing using a 100-minute pumping test. This method was selected because the 1-inch diameter deep well cannot accommodate slug testing. Pumping tests also evaluate a larger portion of the aquifer than slug tests, thus providing a more accurate evaluation of aquifer characteristics. The pump tests at Landfill Areas 1 and 3 were conducted on well LF1DW01 and LF3DW01, respectively. Slug test and 100-minute pump test data are discussed in Sections 3.1.6 and 3.2.6, and presented in Appendix F.

2.2.12 Sample Location Survey

All soil sample locations, monitoring wells, piezometers, dirt mounds, test pits, and other miscellaneous features were surveyed to determine horizontal and vertical map coordinates. Surveying services were provided by GeoMap Technologies, Inc., Tampa, Florida, a professional land surveyor and mapper registered in the state of Florida. Initially, GeoMap established an extensive network of horizontal and vertical control points for Landfill Area 2 and the LFNAS, as a whole. Following the establishment of control points, GeoMap mobilized on four other occasions (August 24 through October 25, 2000, October 30 through November 10, 2000, February 5 through 7, 2001, and September 2002) to survey points at Landfill Areas 1 and 3. Horizontal locations of all sample points, piezometers/wells, and pertinent site features (fence lines, ditches, mounded areas, etc.) were surveyed and referenced to the North American Datum 1983 Geodetic and Florida East, State Plane Coordinates. Vertical elevations including piezometer/monitoring well ground and top of casings elevations were surveyed to the nearest 0.01 feet and were referenced to the 1988 North American Vertical Datum. The ground elevations of all soil sample points were also surveyed. A copy of the survey report and data are provided in Appendix G.

2.2.13 Decontamination Procedures

Decontamination of data acquisition and sampling equipment was performed in accordance with Section 6.1.7 of the SAP (IT, 2000a) to maintain the integrity of the soil and groundwater samples collected during the field investigation.

2.2.13.1 Drilling and Geoprobe Equipment

A decontamination pad was constructed and lined with three layers of at least 6-mil-thick Visqueen® and was bermed on three sides for decontamination of drill rigs and drilling equipment. The decontamination pad was constructed so that the drill rigs could be driven onto the Visqueen® for decontamination, and all decontamination water could be collected and pumped into drums. All drill rigs and equipment were decontaminated when first brought on Base and before each demobilization from the Base.

Geoprobe drive rods were washed in the field using alconox soap followed by a potable water rinse during advancement of the borehole. The soil sampler and drop-out stainless-steel screens for DPT groundwater sampling were decontaminated by washing with alconox soap, followed by a potable water rinse. This was followed with a rinse of pesticide-grade isopropanol and an analyte-free water rinse. The Geoprobe machine was loaded onto a trailer and taken to the central decontamination area and decontaminated using high pressure steam before and after use at the site.

Drilling equipment used for mud rotary and hollow-stem auger such as rods, bits, and augers, were either steam cleaned on a temporary decontamination pad constructed at the site or at the central decontamination area. All down-hole drilling equipment was decontaminated between each borehole and the rig was decontaminated between each site.

2.2.13.2 Sampling and Aquifer Test Equipment

All sampling, well development, and aquifer test equipment was decontaminated prior to use and between samples or tests. Decontamination involved: a potable water rinse, wash with alconox or liquinox, potable water rinse, isopropanol rinse, analyte-free water rinse, air dry, and wrapping the equipment in foil.

2.2.14 Documentation

All field activities were documented on quality control reports during the Phase I and Phase II RIs. Field documentation forms used included field activity daily logs, sample collection logs, HTRW drilling logs, well completion logs, well development/purge logs, and analysis requests/chain-of-custody records. The procedures for identifying and documenting samples collected during the RI are presented in Section 6.3.3 of the SAP (IT, 2000a).

2.2.15 Landfill Gas Monitoring

Landfill Areas 1 and 3 were monitored for gas emissions using a calibrated FID equipped with a charcoal filter and a MSA Five Star Passport. The Passport is calibrated to measure oxygen, carbon monoxide, hydrogen, sulfide, and methane. An FID was also used in conjunction with the Passport to measure ambient air conditions at each monitoring point. The ground surface was monitored for methane emissions by traversing across the surface in a serpentine pattern and holding the instrument approximately 1 to 2 feet above the ground surface.

The subsurface was monitored through the use of barhole probes. The barhole probes were constructed of 1-inch ID PVC pipe with 3/8-inch holes drilled in the lower two feet of the pipe. Twenty-six probes (BP-1 through BP-26) were installed around the perimeter of each landfill to depths of 3 to 4 feet bgs. The location of the borehole probes installed at Landfill Areas 1 an 3 are detailed on Figures 1-3 and 1-4, respectively. The boreholes for the probes were drilled with a hand auger. Once installed in the boreholes, the annulus between the pipe and the borehole wall was backfilled with the native soil to 1 foot bgs. The top of the annulus was sealed with bentonite to prevent possible gas migration out the top of the borehole. The top of the probes were capped with tight slip caps.

2.3 Sample Analysis

Off-site analyses were performed by Accutest Laboratories of Orlando, Florida (Accutest) using EPA methods and meeting the QA/QC requirements described in the referenced SW-846 methods and the LFNAS SAP (IT, 2000a). Accutest analyzed all groundwater samples, except Phase II samples collected from wells LF1MW04 and LF1DW01, for VOCs (EPA 8260B), SVOCs (all samples by EPA 8270C and 20 percent of those by EPA 8310 also), TAL metals (EPA 6010B/7471), TRPH (FL-PRO Method), pesticides (EPA 8081A), and PCBs (EPA 8082). Groundwater samples collected by direct-push were analyzed for VOCs and SVOCs (EPA 8270C and 20 percent of those by EPA 8310 also) only, as a screening tool. Soil samples were analyzed for VOCs, SVOCs, PAHs (20 percent only), TAL metals, TRPHs, pesticides, and PCBs. These methods were chosen because they provide the most comprehensive set of constituents for analysis, which is necessary due to the unknown type of debris at the landfill.

Field-split samples for quality assurance were submitted to Accura Analytical Labs, Inc. of Norcross, Georgia.

2.4 Data Management

The data management process includes all aspects of data review and data validation. All data associated with the LFNAS project was subjected to several evaluations in the laboratory and by

Shaw personnel prior to data release to an end data user. General data management followed the guidance provided in Chapter 8.0 of the SAP (IT, 2000a).

- ***Data Packages.*** The data packages received from the laboratory were reviewed internally by the analyst and a peer or supervisor prior to being sent to the Shaw project chemist for additional project-specific review. In the data review process, the data were compared to the planned objectives in the work plan and QC sample data to evaluate the validity of the results.

- ***Data Validation.*** A thorough evaluation of the data was conducted to determine whether or not the project objectives had been met. Specific issues addressed included precision, accuracy, and representativeness such as duplicate results, matrix spike/matrix spike duplicate, and blank sample results. An evaluation of completeness was performed, and data deficiencies were identified and rectified or documented for the report.

- Data was validated in accordance with the SAP (IT, 2000a). One hundred percent of the data collected were validated. During the data validation process, qualifiers were added to the data. The formal data validation qualifiers used for the project are listed and described in the National Functional Guidelines (EPA, 1993, 1994a). All aspects of validation as defined in the National Functional Guidelines were evaluated for each set of data collected. The QA/QC guidance defined in EPA's *Test Methods for Evaluating Solid Waste* (EPA SW-846) (EPA, 1986a) were used unless more stringent criteria are defined in the LFNAS SAP.

A summary of all analytical data and the data quality control summary report are provided in Appendix I.

2.5 Investigation-Derived Waste

Phase I Investigation-Derived Waste. The RI at the Landfill Areas 1 and 3 generated development and purge water, soil cuttings, drilling mud, decontamination liquids, and other investigation-derived waste (IDW). A total of 21 drums of IDW (2 drill cuttings, 9 development/purge water, 6 decontamination water, and 4 personal protective equipment) were generated during the Phase I fieldwork at the Landfill Areas 1 and 3. The decontamination water was drummed with other decontamination water for disposal on a site-wide basis. IDW of different media were stored separately in steel U.S. Department of Transportation-approved 55-gallon drums. Each container was marked showing the date of collection, the area, the nature of the waste (e.g., solid or liquid), and the name and telephone number of the contact person. The location and contents of drums were tracked on an IDW/drum inventory log until disposal could

be arranged. All drums were sealed to prevent leakage or introduction of contamination from external sources.

First, it was determined if the IDW generated was a Resource Conservation and Recovery Act (RCRA)-listed waste or a characteristic waste (hazardous waste as defined in Title 40 Code of Federal Regulations Part 261) based on knowledge of the generating process. As there was no definitive documentation of the contaminants released or their date of release, the IDW is not classified as a listed hazardous waste. Since knowledge of the generating process was not sufficient to allow characterization of the IDW, laboratory analyses were performed on the waste. A composite sample from the solid IDW was collected for analysis. Liquid IDW was characterized as needed for decontamination fluids or on a per well basis for well purge and development water.

Following the receipt of analytical results, the data were reviewed by one of Shaw's regulatory specialists for determination of disposal options. A memo documenting the results and including recommendations for disposal was prepared and submitted to the USACE for review and approval. The results indicated that none of the chemicals detected in the IDW exceeded toxicity characteristic leaching procedure (TCLP) limits. However, to avoid future concerns from the site owner and regulatory agency, it was decided that the waste should be transported off site for disposal at a Subtitle D facility as a nonhazardous waste. This decision was made after analysis of the data and consultation with the FDEP, the USACE, and the Clay County Port.

Shaw contracted U.S. Waste Logistics, Inc. (US Waste) of Green Cove Springs, Florida, to provide transport and disposal of the IDW at LFNAS. Prior to drum pickup, Shaw provided U.S. Waste with a waste profile document. From the waste profile, U.S. Waste provided nonhazardous waste manifests for the IDW. The manifests were reviewed and signed by a USACE representative prior to drum pickup. Drums were retrieved by U.S. Waste on March 21-22, 2001, and June 7, 2001. All drums were transported to the U.S. Waste facility for treatment and then sent to the Chesser Island Road Landfill near Folkston, Georgia, for disposal (Appendix J).

Phase II Investigation-Derived Waste. The Phase II RI generated a total of 8 drums of IDW (3 drums development/ purge water, 4 drums soil cuttings, and 1 drum decontamination water). The same procedures are being followed as were used for disposal of the drums from Phase I. Based on process knowledge, historical data, and review of Phase II analytical data, all drums required sampling for waste characterization. Sampling was conducted on March 20,

2003. The decontamination water drum was analyzed for VOCs, TRPH, PCBs, and RCRA metals. Because of the presence of high concentrations of VOCs in groundwater, samples were collected from each of the purge water drums and analyzed for VOCs only. A composite soil sample was collected from the four soil drums analyzed for RCRA metals, TCLP organics, TRPH, and PCBs. Analytical results from the drum sampling were received on April 18, 2003. All drums generated during the Phase II investigations, including those from the underground storage tank areas and HTRW sites will be removed upon USACE, FDEP, and Clay County Port approval.

3.0 Site Characteristics

3.1 Landfill Area 1

3.1.1 Site Description

Landfill Area 1 covers approximately 6.2 acres of land near the southeastern corner of the LFNAS property. The landfill is bordered on the north by a fence line, Wildwood Road and a drainage swale paralleling Wildwood Road to the north; east by a fence line, stream bed, and a cypress wetland area (Three Mile Swamp); south by Three Mile Swamp; and west by a low, flat, wooded area that contains standing water during the wet season and connects with Three Mile Swamp (Figure 1-3).

Currently, the landfill is completely covered with vegetation of different types depending upon the surface cover. The surface of the eastern one-half to two-thirds of the landfill is generally mounded with abundant debris approximately 8 to 15 feet above the surrounding topography. In this portion of the landfill, debris is generally covered with a thin layer of soil, which supports mostly grasses, scrub vegetation, and few trees. Various types of debris such as concrete and brush/timber piles are present on the surface or are visible through the soil cover.

An entrance gate is present at the northeast corner of the landfill. From the gate, a grassy access path can be followed that extends south over the mounded surface and down a grassy ramp that ends at the wetland. The eastern edge of the landfill slopes downward toward the wetland area and is dominated by cypress trees and low vegetation common to wetland areas in the southeastern U.S. The eastern portion of the landfill contains debris such as concrete culverts, drummed materials, and reportedly power poles.

The southern edge of the landfill is low and relatively flat as the ground surface slopes downward from the high central portion of the landfill. The southeastern and southwestern edges of the landfill are covered mostly with grasses and low vegetation common near wetland areas.

The south-central part of the landfill is densely vegetated with cypress trees and vines. The central and west-central portions of the landfill are dominated by thick vegetation such as trees, brush, and scrub, and are essentially inaccessible by foot and most equipment. Mounded debris is common here including concrete, fiberglass materials, rusted drums (some labeled "acetone"), and other miscellaneous debris. The western portion of the landfill is fairly flat and contains a

dense canopy of trees, but is accessible. Surface debris is common here, especially to the south. The northern edge of the landfill slopes downward from the center to Wildwood Road. The area has little surface debris visible. Most all of the debris observed at the surface is known to be related to post-DOD activities.

3.1.2 Site Historical and Current Use

PHR, a subsidiary of Shaw, conducted a thorough records search of the entire LFNAS facility in 2000. During the search, various documents and maps were obtained and reviewed to determine the past history of the facility and individual sites at LFNAS. The findings are included in the Historical Operations Summary Report (PHR, 2001). Shaw supplemented information from the PHR report with an aerial photograph review and interviews with LFNAS employees.

No documents were discovered which discussed the specific disposal practices at Landfill Area 1. The landfill was reportedly used primarily during and immediately after World War II. Aerial photographs indicate the landfill area was first actively used between 1947 and 1948 by the U.S. Navy and continued until closure of the Base in 1963. A 1950 Base map of the LFNAS indicates the area now known as Landfill Area 1 denoted as the "Sanitary Yard Dump". The sanitary yard dump was depicted inside a "hog proof fence" on a December 1951 map measuring 500-feet north-south by 700-feet east-west. Other maps including the 1953 Base map and 1958 index of structures indicate the presence of the landfill. A 1961 map shows the fence but does not list the landfill.

Shaw interviewed a civilian who worked at LFNAS during the time it was used by the Navy. According to the former employee, surplus materials and waste from the ships were either burned in an open-air incinerator located just south of (behind) the Small Arms Range berm or disposed at the "main dump" (Landfill Area 1) or the small dump located between the incinerator and Landfill Area 1. Another former employee interviewed by Shaw indicated that all waste materials from the electric shop were placed either in the main dump or small dump. Interviews and aerial photos indicate dumping was limited to the eastern 1/3 (mostly the northeast corner) of the landfill area. Some of the known disposal material reportedly consisted of compost and creosote-treated power poles (visible on the eastern edge of the landfill). Burning of materials was also conducted on the landfill during the operational period of LFNAS.

Later aerial photographs confirm use of the landfill for disposal following closure of LFNAS until approximately 1990. Most activity, including clearing of the landfill, disposal and burial of

debris, placement of fill material, and grading, was observed during the 1970s and 1980s. In the 1980s the landfill was used to dump demolition debris such as concrete and fiberglass materials.

3.1.2.1 Aerial Photograph Interpretation

A thorough review of available aerial photographs was conducted to determine the history and use of Landfill Area 1. The first available aerial photograph obtained for the LFNAS area is September 30, 1940 and shows the LFNAS in the beginning stages of construction. The latest available aerial photograph obtained to indicate current conditions was taken by the Florida Department of Transportation on February 6, 2001. The following is a list of pertinent aerial photographs reviewed and the site conditions observed on those dates.

May 20, 1944. The area later occupied by Landfill Area 1 is still undeveloped and appears vegetated. The Small Arms Range is present to the north of the future Landfill Area 1. The Base perimeter road (Wildwood Road) is not yet present (Figure 3-1).

February 17, 1947. No activity is evident at the future location of Landfill Area 1. This is the last aerial photograph available prior to development of the landfill.

March 29, 1948. This is the first aerial photo showing activity at Landfill Area 1. The landfill appears as a circular-shaped cleared area that has a diameter of approximately 150 feet (Figure 3-2). A drainage ditch and dirt access road has been completed from the north end of Runway 5-23 south to the landfill location.

August 29, 1952. Landfill Area 1 is still circular in shape, generally unchanged from 1948. However, a dirt road or path extends from the southern portion of the landfill to the west beyond where the landfill has been developed.

October 21, 1954. In this oblique aerial photograph, Landfill Area 1 appears to have been expanded to the west from its origin. The cleared area has now approximately doubled in size since 1952.

November 19, 1958. This is a clear aerial photograph and stereo pair showing the landfill has been increased to its maximum western extent (Figure 3-3). The landfill is somewhat mottled in appearance, which seems to indicate both cleared and vegetated areas. Relief indicates some debris is present. The landfill is bordered on the west, south, and east by heavily wooded land

occupied by Three Mile Swamp. This is the last aerial photograph available prior to closure of the LFNAS (Figure 3-3).

November 9, 1969. In this photograph, vegetation on Landfill Area 1 has begun to recover since 1958 and closure of the Base. Only the eastern half of the landfill appears recently disturbed. The landfill surface appears hummocky, indicating disposal has occurred since 1958.

April 25, 1971. Here, all of Landfill Area 1 has been cleared or disturbed except the extreme western edge and north-central to northwest corner (Figure 3-4). The eastern half and southwestern corner appear hummocky due to dumped debris. A small dirt road or path extends to the west across the landfill.

November 15, 1975. The landfill debris has apparently been covered with fill material and graded, as it is very smooth in appearance. All but the extreme northwest corner has been cleared or disturbed. The landfill area has expanded to the south into the swamp.

November 26, 1979. The landfill is similar in appearance to 1975, except more debris is present and is visible on the eastern third of the site and southwest corner.

January 29, 1981. The landfill again looks cleared of fill covered with abundant debris along the southeast and southwest corners. The landfill is being expanded to the south into the swamp.

March 22, 1984. Landfill Area 1 is similar in appearance to 1981, except the east side is more rounded as it has been expanded farther into the wetland area. Abundant debris is visible.

February 10, 1987. The landfill has been expanded to its maximum width as it appears today (Figure 3-5). Abundant mounded debris is visible throughout the landfill except on the western edge, southwest, and northwest corners, which are wooded.

November 12, 1990. The landfill is similar to 1987, but is beginning to become overgrown by vegetation. No new disposal is apparent.

April 12, 1993. Landfill Area 1 is still hummocky, but overall smoother in appearance. The high central portion of the landfill is visible, as is the debris embankment and low, flat southern and eastern sides. The west end of the landfill is becoming overgrown by trees.

February 6, 2001. This aerial photograph is the most recent reviewed (Figure 3-6). The landfill is becoming overgrown with vegetation on the eastern edge, southern edge, and western half. The dirt access road/path seen today is visible from the entrance gate at the northeast corner south to near the wetland area.

3.1.2.2 Summary of Photograph Interpretation

Based on a review of historical aerial photographs, the first activity in the vicinity of Landfill Area 1 was observed in 1948, with the initial clearing of the northeast corner of the site. The previous aerial photo reviewed (February 17, 1947) showed no activity at the present Landfill Area 1 location. Activity at the landfill continued through the active base period, as documented through the 1958 aerial photograph. Most activity from 1948 to 1952 occurred in the northeast corner of the present landfill footprint. A previous employee at LFNAS indicated that disposal during the LFNAS operational period occurred primarily on the eastern half of the present landfill area. Expansion of the cleared area was observed in 1954 and 1958. The western extent of the disposal and burial area could not be determined by the photographs.

Several periods of activity were documented at Landfill Area 1 by the aerial photographs after LFNAS was no longer operated by the DOD. Clearing, disposal, burial, placement of fill, and grading were all observed during the 1970s and 1980s.

3.1.3 Test Pit Results

A total of four test pits were excavated in an attempt to determine the western boundary of the landfill area and search for debris burial areas (Figure 1-3). The northern boundary of the landfill never extended north of Wildwood Road. The eastern and southern boundaries are distinct due to debris present at the surface and the elevated border with the wetland area. The western boundary was targeted for test pit excavations, as there is no obvious indication of the landfill edge other than aerial photos. The western boundary was estimated based on the size of the landfill at its peak. The test pits were then placed east and west of the estimated boundary. The test pits were excavated below the static water table to a depth of 5 to 7 feet bgs. Soil encountered was generally a black organic-rich soil at the surface grading into a tan, brown, or gray clayey sand with shell fragments. None of the test pits encountered any debris. Though no debris was encountered in the test pits, the western boundary was estimated to be just east of the test pits due to the presence of surface debris.

3.1.4 Site Topography and Drainage

The topography of Landfill Area 1 varies depending upon location. The eastern and southern edges and western portion of the landfill are relatively flat, with little relief. Just inside the eastern and southeastern boundaries, debris is mounded approximately 10 feet higher than the surrounding land surface for a distance of approximately 100 feet. The landfill surface slopes upward as you traverse toward the center. The maximum relief is approximately 12 feet from the top of the landfill to the surrounding wetland areas. The elevation of the site averages approximately 10 feet above msl, with a high elevation of approximately 20 feet above msl.

Drainage for Landfill Area 1 occurs in a radial pattern away from the high central part of the landfill. Three Mile Swamp accepts runoff from the southern, eastern, and to a lesser extent, the western portions of the landfill. Due to the relief of the site, rainwater generally does not pond on the surface. The exception is the western portion of the landfill which collects water during heavy rain events. During the wet season, the western part of the landfill contains standing water that often merges with Three Mile Swamp. Runoff flowing north from the top of the landfill drains into the drainage ditch located just north of Wildwood Road. The drainage swale and portion of Three Mile Swamp that borders the landfill generally contain water for approximately 8 months of the year.

3.1.5 Site Geology

The subsurface geology was determined by the installation of piezometers, monitoring wells, and lithologic borings. A total of six monitoring wells (5 shallow and 1 deep) were installed at the site during the RI field work. Soil borings were drilled to a maximum depth of 40 feet bgs adjacent to each piezometer location to determine the subsurface lithology and determine if a continuous confining unit was present beneath the site. Lithologic information obtained during the drilling activities was used to construct cross-sections for the site. Figure 3-7 indicates the locations of the cross-sections. Cross-sections oriented west to east along line A-A' and south to north along line B-B' are presented as Figures 3-8 and 3-9, respectively.

All borings encountered undifferentiated sediments of sand, silty sand, clayey sand, and some clay. The different sediment types formed irregular and discontinuous lenses found at the perimeter of the landfill. Several borings such as LF1PZ06, LF1PZ05, and LF1PZ03 encountered clay layers; however, these units appear to represent discontinuous lenses. Therefore, there is no true confining unit present at depth. Sand zones were typically gray to brown to greenish-gray-colored, fine grained, and well sorted with mica or phosphate. Deep lithologic borings were drilled as close as possible to the edge of the landfill due to the uncertain

nature of possible debris types. On the south and east sides of the landfill, borings had to be placed on the landfill due to the surrounding wetland area. Landfill material such as fiberglass, paper, and metal was encountered to depths of 6 to 8 feet bgs in borings located along the southern edge of the site (LF1MW04, -MW05, LF1DW01, and LF1PZ04.)

In the deep well, a surface casing was set at 20 feet in a clayey sand to reduce the possibility of cross contamination of the aquifer above and below the clay layer. This unit represents a localized aquitard. Below 26 feet, alternating layers of sand and clayey sand units were encountered to the total depth of the deep well.

The grain size analysis of ten surface soil samples indicated that an average of 86.7 percent of soil collected was finer than 0.425 millimeter (mm) (#40 sieve) and 23.4 percent was finer than 0.149 mm (#100 sieve). Therefore, the surface soil is predominantly a poorly sorted fine to medium grained sand, with a small percentage of coarse and very fine-grained sand.

Geotechnical parameters such as bulk soil density, total soil porosity, air-filled and water-filled porosity, particle density and fractional organic carbon (f_{oc}) were also reported for five vadose subsurface soil samples. In addition, two shallow and one deep saturated soil samples were analyzed for grain size, moisture, bulk density, porosity, and vertical conductivity. A copy of the geotechnical reports can be reviewed in Appendix E.

3.1.6 Site Hydrogeology

Five new shallow wells and one deep well were installed in the vicinity of Landfill Area 1 during the RI. The five new shallow wells were screened from approximately 2 to 12 feet bgs. The pre-existing shallow wells are screened from approximately 4 to 14 feet bgs. The deep monitoring well is located adjacent to LF1MW04 and is screened from 24 to 29 feet bgs.

Groundwater in the upper portion of the aquifer is contained under water table conditions in undifferentiated sand, silt, and clayey sand layers. No horizontally continuous confining clay layer was encountered to 40 feet bgs to separate the surficial aquifer from lower water-bearing zones.

The water table at Landfill Area 1 was encountered at approximate depths of 1 to 4 feet bgs during the drilling activities and subsequent water level monitoring events. Groundwater levels were measured on several occasions during the RI, as described in Section 2.2.10. Groundwater elevations are summarized in Table 3-1. Figures 3-10, 3-11, and 3-12 present the water table configuration on February 12 and July 11, 2001 and October 8, 2002, respectively. As indicated

from the water table elevations, groundwater generally flows away from the high central portion of the landfill in a radial pattern. The average gradient across the site is approximately 0.00111 feet/foot.

Aquifer testing in the form of slug tests were performed to determine hydraulic properties of the aquifer in the immediate vicinity of the test wells. Rising head slug tests were conducted in the two pre-existing 2-inch diameter monitoring wells (LF1W-6-86 and LF1W-7-86) in January 2001. At least two tests were run in each well for QC purposes. Average conductivities of 1.02 x 10^{-3} feet/min and 9.78 x 10^{-4} feet/min were calculated for wells LF1W-6-86 and LF1W-7-86, respectively. A groundwater flow velocity of 0.008 feet/day or 2.9 feet/year was calculated based on an average hydraulic conductivity of 1.00 x 10^{-3} feet/min.

3.2 Landfill Area 3

3.2.1 Site Description

Landfill Area 3 covers approximately 7.4 acres of land at the southern edge of the LFNAS property. Previous reports by others estimated the landfill size at 4 acres. However, the estimated size has increased based on revised site boundaries. The landfill is bounded on the north, west, and east by a thick cover of trees. Three Mile Swamp lies approximately 400 feet east of the landfill. The southern edge of Landfill Area 3 is bordered by a dirt access road that connects CR 209 with Wildwood Road and then a wooded area. Farther west past the wooded area is CR 209. The remnants of wooden posts (6 x 6) from a previous fence are located around the perimeter of the landfill (Figure 1-4).

Currently, the landfill is heavily wooded with trees and some undergrowth. In February and March 2001, the eastern edge of Landfill Area 3 was cleared of larger timber from the bend in the access road to the northeast. The removal of timber in these areas has provided more sunlight which allowed the growth of grasses and thick scrub vegetation. Portions of the landfill where this has occurred are now nearly impenetrable, as is the northeast corner of the landfill where cutting debris has been piled.

Overall, the landfill area is relatively flat; however, three mounded areas are located within the estimated boundaries of Landfill Area 3. Two of the mounds are located near well LF3W-5-86. The third mound is located just west of well LF3MW04. The mounds are approximately 3 to 5 feet high and their origin is unknown. Scattered debris including bottles, cans, and miscellaneous metal items are observed at the surface from the south-central portion of the

landfill to the east and northeast. From the surface, there are no other indications that a landfill is present. It is likely that the scattered debris observed at the surface today is the result of random post-DOD dumping.

3.2.2 Site Historical and Current Use

No documents were discovered which discussed the specific use of Landfill Area 3 or possible disposal practices. However, the landfill was reportedly used to dispose of refuse, domestic waste, and medical waste. Following the closure of the Base in 1963, there was some indication of disturbance of the ground surface on a 1969 aerial photograph, slightly north and east of the main disposal area shown on the 1958 aerial. The scattered surface debris present such as bottles and cans likely indicates minor, random dumping that often occurs in remote, wooded areas.

3.2.2.1 Aerial Photograph Interpretation

A thorough review of available aerial photographs was conducted to determine the history and use of Landfill Area 3. The following is a list of pertinent aerial photographs reviewed and the site conditions observed at Landfill Area 3 on those dates.

January 5, 1943. There is no activity in the area later occupied by Landfill Area 3 or in surrounding areas.

May 20, 1944. This is the first indication of activity in the vicinity of what will become Landfill Area 3 (Figure 3-13). A double dirt road has been cut from CR 209 northeast to where the current road to Landfill Area 3 bends. At that point, the northern road turns to the north-northeast and the southern road turns to the south and then turns northeast. Just east of the road split is what appears to be some type of industrial plant or activity. The quality of the aerial does not allow for a better interpretation and no information exists. This is the location where elevated concentrations of chlorinated hydrocarbons exist in the groundwater today. A linear feature, possibly a road or ditch that begins at CR 209 and extends north-northeast is visible northwest of the future Landfill Area 3 location.

March 29, 1948. The area later occupied by Landfill Area 3 has not been developed (Figure 3-14). The activity and structures evident in the 1944 photo have disappeared. The double road described in the 1944 aerial photo has become overgrown in places, with the southern road nearly obscured. The area south of the southern road and west of the split has been cleared.

August 3, 1951. The area north of the northern access road is now outlined with a path or fence line. The southern access road is no longer visible. Some minor clearing activity or ground disturbance is observed along the southern landfill border with the access road.

August 20, 1952. This aerial photograph shows the outline of Landfill Area 3 cleared of most vegetation. The southern portion has been scraped. The circular-shaped feature that is now occupied by a drainage swale and pond in the south-central portion of the landfill is visible.

October 21, 1954. Here, Landfill Area 3 appears mature like the 1958 aerial, but is less developed.

November 19, 1958. Clear photograph and stereo pair indicating Landfill Area 3 at the height of its development (Figure 3-15). The landfill is bound by a distinct unpaved perimeter road. Inside the road is a large disturbed area that appears as bright white. This is apparently the main portion of the landfill that was used for disposal purposes. A relatively undisturbed area is present south of the main disposal area. This area also appears as early as 1952 and presently correlates to a drainage swale and pond. Several roads are seen entering and leaving the landfill on the northeast side. What appears to be a trench and possibly a mound are present on the northeastern part of Landfill Area 3. County Road 209 has been re-routed farther west than it appears in the 1948 aerial photo. The linear feature observed on the 1944 and 1948 photos is mostly obscured, except for the south-southwestern end, where it crosses the former CR 209 and intersects with the new CR 209. This feature now appears to contain water.

November 9, 1969. According to this aerial photograph and stereo pairs for this date, vegetation has begun to recover on portions of the landfill (Figure 3-16). The northeastern part of the landfill has a hummocky appearance due to debris and may have been disturbed since 1958. All other features visible on the 1958 aerial are seen here. The area southeast of the access road previously occupied by the unknown plant or facility appears cleared of most vegetation and is circular in shape.

November 15, 1975. Landfill Area 3 is becoming overgrown with vegetation, especially on the west side and south of the circular ditch/swale; it has a very similar appearance to the 1969 aerial.

November 26, 1979. Landfill Area 3 is completely overgrown except the northeastern portion that was covered with debris.

November 12, 1990. Landfill Area 3 appears completely wooded. The access road between CR 209 and Wildwood Road is almost completely obscured by vegetation.

February 6, 2001. Landfill Area 3 is unchanged from the 1990 aerial photograph (Figure 3-17). It is heavily wooded, with little undergrowth. Shortly after the 2001 aerial photo was generated, Clay County Port cleared part of the landfill by removing some of the large trees. As a result, the cleared areas have become overgrown with grass, weeds, and scrub vegetation that grow readily with more sunlight.

3.2.2.2 Summary of Photograph Interpretation

Based on a review of historical aerial photographs, the first landfill-related activity in the vicinity of Landfill Area 3 was observed by aerial photo in 1951, with the site being cleared. Prior to that time, activity directly adjacent to the landfill was noticed as early as 1944. Activity at the landfill continued through the active Base period, as documented through 1958.

Some activity at Landfill Area 3 was documented by a change between the 1958 and 1969 aerial photo. The 1969 aerial indicates debris apparently present on the northeastern portion of the landfill. This activity may have occurred after closure of the LFNAS in 1963, but this cannot be confirmed.

3.2.3 Test Pit Results

A total of twenty-five test pits were excavated in an attempt to determine the boundaries of Landfill Area 3 and search for debris burial areas (Figure 1-4). The test pits were excavated to depths ranging from 2 to 7 feet bgs, photo documented, and then backfilled. Initially, test pits were located to target perimeter areas of the presumed landfill. These test pits were generally excavated to the static water table only. When perimeter locations yielded no debris, additional test pits were placed within the landfill interior, to target burial areas. These test pits were placed based on the 1958 aerial photograph. Most test pits encountered an organic-rich sand or silt grading into sand or clayey sand at depth. Test Pits 9, 14, 15, and 23 all encountered various types of debris and waste, primarily below the static water table. These test pits confirm that waste burial occurred in disturbed areas identified on the 1958 aerial photo. Test Pit 9 encountered gas masks, hoses, and cartridges, glass, metal, newspaper, trash cans, mufflers, and other miscellaneous debris below the water table at depths of 3 to 6 feet bgs. Metal debris and a petroleum/grease odor were noticed below the water table from 3 to 6 feet bgs in Test Pit 14.

Glass and metal debris were discovered in Test Pits 15 and 23. No other test pits encountered debris of any type.

3.2.4 Site Topography and Drainage

The topography of Landfill Area 3 is relatively flat with a slight slope toward Three Mile Swamp to the east and northeast. The elevation of the site averages approximately 15 feet above msl. Three mounded areas are present in the central portion of the landfill and along the eastern boundary. These are isolated mounds that range in height from 3 to 6 feet above the surrounding ground surface.

Drainage for Landfill Area 3 is generally toward the east into Three Mile Swamp, which is the nearest downgradient receptor. Locally, water drains into the drainage swale/ditch and pond in the south-central portion of the landfill. The swale is generally dry except during the rainy season or during heavy rain events. The ponds on-site generally remain wet year round.

3.2.5 Site Geology

The subsurface geology was determined by the installation of piezometers, monitoring wells, and lithologic borings. A total of eight monitoring wells (6 shallow and 2 deep) were installed at the site during the RI field work. Monitoring wells LF3MW01 and –MW02 were converted from LF3PZ01 and –PZ05, respectively. Soil borings were drilled to a maximum depth of 40 feet bgs adjacent to each piezometer location to determine the subsurface lithology and determine if a continuous confining unit was present beneath the site. Lithologic information obtained during the drilling activities and from the two pre-existing monitoring wells was used to construct cross-sections for the site. Figure 3-18 indicates the locations of the cross-sections. Cross-sections oriented west to east, then north along line A-A' and northwest to southeast along line B-B' are presented as Figures 3-19 and 3-20, respectively.

All borings encountered undifferentiated sediments of sand, silty sand, clayey sand, and minor clay. The different sediment types formed irregular and discontinuous lenses found at the perimeter of the landfill. The predominant sediment type observed was a fine-medium grained, white to greenish gray to brown sand. The sand units ranged from poorly to well sorted. Minor lenses of silt and clay were encountered, but they were not continuous across the site. Deep lithologic borings were generally not drilled in the interior of the presumed landfill area due to the uncertain nature of possible debris types. Landfill material was encountered to a depth of approximately 6 feet bgs in LF3W-5-86.

Surface casings were set in deep wells LF3DW01 and LF3DW02 at 25 and 23.8 feet, respectively, to reduce the potential for cross contamination of lower zones within the surficial aquifer. The casings were set in a sand unit, as no clay layer was encountered at the appropriate depth interval.

The grain size analysis of ten surface soil samples indicated that 96.6 percent of soil collected was finer than 0.425 millimeter (mm) (#40 sieve) and 35.7 percent was finer than 0.149 mm (#100 sieve). Therefore, the surface soil is predominantly poorly sorted, fine-grained sand, with a small percentage of medium and very fine-grained sand and silt. The grain size of the shallow saturated zone was determined based on two soil samples collected from 6 to 10 feet bgs. The soil collected averaged 98.5 percent finer than a #40 sieve with only 25.4 percent finer than #100 indicating moderately sorted fine-grained sand. The saturated soil sample collected from 26 to 31 feet bgs in the aquifer is predominantly fine grained sand, but also has grain sizes ranging from medium to silt and is poorly sorted.

Geotechnical parameters such as bulk soil density, total soil porosity, air-filled and water-filled porosity, particle density and fractional organic carbon (foc) were also reported for five vadose subsurface soil samples. In addition, two shallow and one deep saturated soil samples were analyzed for grain size, moisture, bulk density, porosity, and vertical conductivity. A copy of the geotechnical reports can be reviewed in Appendix E.

3.2.6 Site Hydrogeology

Six new shallow wells and two deep wells were installed in the vicinity of Landfill Area 3 during the RI. The six new shallow wells were screened from approximately 2 to 12 feet bgs. The pre-existing shallow wells are screened from approximately 9 to 19 feet (LF3W-4-86) and 5 to 15 feet (LF3W-5-86) bgs. Deep monitoring well LF3DW01 is located adjacent to LF3MW04 and is screened from 29 to 34 feet bgs. A second deep monitoring well was installed downgradient of the chlorinated solvent detection at the southeast corner of the landfill.

Groundwater in the upper portion of the aquifer is contained under water table conditions in undifferentiated sand, silt, and clayey sand layers. No clay confining unit is present in the upper 40 feet to separate the surficial aquifer from lower water-bearing zones.

The water table at Landfill Area 3 was encountered at approximate depths ranging from 1 to 4 feet bgs during the drilling activities. Groundwater levels were measured on several occasions during the RI, as described in Section 2.2.10. Groundwater elevations are summarized in Table 3-1. Figures 3-21, 3-22, and 3-23 present the water table configuration on February 13 and July

11, 2001 and October 8, 2002, respectively. As indicated from the water table elevations, groundwater generally flows toward the east-northeast gradient across the site based on the February 13, 2001 water data is approximately 0.00495 feet/foot.

Aquifer testing in the form of slug testing was performed to determine hydraulic properties of the shallow aquifer in the immediate vicinity of the test wells. Rising head slug tests were conducted in the two pre-existing 2-inch diameter monitoring wells (LFW-4-86 and LF3W-5-86) in January 2001. At least two tests were run in each well for QC purposes. Average conductivities of 3.04×10^{-5} feet/min and 3.82×10^{-4} feet/min were calculated for wells LFW3-4-86 and LF3W-5-86, respectively. A groundwater flow velocity of 0.004 feet/day or 1.4 feet/year was calculated based on average hydraulic conductivity of 2.06×10^{-4} feet/min.

A 100-min pumping and recovery test was conducted for deep well LF3DW01 to determine the characteristics of the deeper portion of the shallow aquifer. The average conductivity and transmissivity values for the drawdown and recovery test were 0.0026 feet/min and 8,976.5 gallons per day/foot. Aquifer test data and interpretation is found in Appendix F.

The porosity of the shallow and deep saturated aquifer zones was measured in the laboratory from two undisturbed soil samples collected at 6 to 10 feet and one sample collected from 26 to 31 feet bgs. The total effective porosity measured ranges from 38.0 to 44.7 percent in the shallow zone and 41.8 percent in the deep saturated zone. A copy of the geotechnical report is included in Appendix E.

4.0 Sample Analytical Results

The following sections of this report provide the results of the environmental sampling conducted during the RI. Surface and subsurface soil, sediment, groundwater, and surface water samples were collected at both landfill areas. All surface soil samples were collected from the 0 to 1-foot interval. Subsurface soil samples were generally collected from the 1 to 3-foot interval.

All analytical results were compared to the applicable standards in accordance with FDEP regulations (FDEP, 1999). The soil results were compared to the lower SCTL of the direct-exposure residential standard or the leachability criteria based on groundwater. Groundwater results were compared to groundwater criteria, and surface water analytical results were compared to the fresh surface water criteria of the groundwater cleanup target levels (GCTL). Results from sediment sampling were evaluated against the toxic effect level (TEL) of the Sediment Quality Assessment Guidelines (SQAG). Metals results from soil and groundwater samples were also compared to background concentrations (Shaw, 2004). Detections of inorganic analytes classified as nutrients, such as calcium, magnesium, and potassium, are presented in the summary tables, however, are not included of maps or in discussions.

4.1 Landfill Area 1

4.1.1 Surface Soil Analytical Results

Surface soil samples were collected from twenty borings in the vicinity of Landfill Area 1. As described previously, the soil samples were analyzed for VOCs, SVOCs, TRPH, TAL metals, PCBs, pesticides, and PAHs (20 percent of the samples).

Organic compounds detected in surface soil are summarized in Table 4-1. The spatial distribution of organic compounds is shown on Figure 4-1. Only three VOCs were detected in site surface soil, none of which exceeded SCTLs.

A total of thirteen SVOCs were detected in nine samples, five of which exceeded their respective SCTLs and background concentrations. One or more of the five compounds, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene, were detected above SCTLs and background in four samples. Concentrations ranged from an estimated 0.816J milligrams per kilogram (mg/kg) of dibenzo(a,h)anthracene in LF1SB06 to an estimated 6.27 mg/kg of indeno(1,2,3-cd)pyrene in

LF1SB04. The sample locations (LF1SB02, LF1SB04, LF1SB06, and LF1SS05) are scattered around the site and thus, there is no discernable pattern to SVOC contamination (Figure 4-1).

PCBs were detected in 17 of 20 surface soil samples. Aroclor 1254 was detected in two samples, LF1SB03 (0.0353J mg/kg) and LF1SB04 (1.42 mg/kg) with the concentration at SB04 exceeding the SCTL. Aroclor 1260 was detected in 15 surface soil samples at concentrations ranging from an estimated 0.0137 mg/kg at LF1SS08 to 2.3 mg/kg at LF1SB05. Three samples (LF1SB05, LF1SS01, and LF1SS03) contained Aroclor 1260 at concentrations exceeding SCTLs. As with the SVOCs, no discernable pattern can be seen in the distribution of exceedences (Figure 4-1).

Seven pesticides were detected in twelve surface soil samples. Dieldrin was the only pesticide detected above the SCTL with concentrations of 0.0071J mg/kg at LF1SS01, 0.0083J mg/kg at LF1SS03, and 0.193 mg/kg and LF1SS05.

TRPH was detected in each sample with concentrations ranging from 13.1 to 462 mg/kg. Three samples, LF1SB04, LF1SB05, and LF1SS03, contained TRPH above the SCTL.

Metals were detected above SCTLs and background in eight samples (Table 4-2). Analytes detected above background and SCTLs included antimony, arsenic, barium, chromium, copper, iron, lead, and vanadium. In addition, eleven other analytes were detected above background but below SCTLs. The most common analyte exceeding the screening values was arsenic with concentrations ranging from 0.92 to 12.4 mg/kg. Other maximum concentrations included antimony at an estimated 150J mg/kg, barium at an estimated of 1,990J mg/kg, and iron at 55,700 mg/kg. Samples collected at LF1SB04, LF1SB05, and LF1SS03 contained the most analytes above background and SCTLs. These samples were located in the southeastern portion of the site (Figure 4-2).

4.1.2 Subsurface Soil Analytical Results

Subsurface soil samples were collected from eight borings in the vicinity of Landfill Area 1. As described previously, the soil samples were analyzed for VOCs, SVOCs, TRPH, PCBs, pesticides, and TAL metals. The analytical results are summarized in Tables 4-3 and 4-4.

No VOCs were detected in subsurface soil samples above SCTLs. Five SVOCs (benzo[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, dibenz[a,h]anthracene, and indeno[1,2,3-cd]pyrene) were detected above SCTLs at LF1SB06 at concentrations ranging from an estimated 1.82 mg/kg of dibenzo(a,h)anthracene to 4.16 mg/kg of benzo(b)fluoranthene.

One PCB and one pesticide were also detected above SCTLs. Aroclor 1260 was detected in LF1SB05 at 5 mg/kg. Dieldrin was detected in LF1SB04 at an estimated 0.0098 and 0.0116 (duplicate) mg/kg.

Concentrations of TRPH ranged 7.78 at LF1SB01 to an estimate 554J mg/kg in LF1SB04. Three soil boring samples contained TRPH above the SCTL, including LF1SB04, LF1SB05, and LF1SB06.

A total of sixteen TAL metals were detected in subsurface soil above either SCTLs or background with five metals exceeding SCTLs and background including antimony, arsenic, barium, chromium, and copper. Concentrations of those analytes exceeding both screening values ranged from 1.2 mg/kg of arsenic at SB06 to an estimated 755J mg/kg at SB05.

4.1.3 Groundwater Analytical Results

4.1.3.1 Direct-Push Results

Prior to installation of the piezometers and permanent monitoring wells, groundwater samples were collected at selected intervals by direct-push from six locations surrounding the landfill. At each location, groundwater samples were collected from approximate intervals of 20 and 40 feet bgs. The samples were analyzed for VOCs and SVOCs to evaluate water quality at the perimeter of Landfill Area 1 and to help determine proper placement and depths of the permanent wells. Following direct-push sampling, piezometers were installed directly adjacent to the six groundwater sample locations.

The analytical results indicated detectable concentrations of bis(2-ethylhexyl)phthalate and fluoranthene. bis(2-Ethylhexyl)phthalate exceeded the Florida GCTL in four samples with concentrations ranging from an estimated 6.6J to 10.5 μg/L. Table 4-5 lists all organic compounds detected during the direct-push groundwater sampling event. Figure 4-3 illustrates the direct-push sample locations and concentrations of organic compounds detected.

4.1.3.2 Monitoring Well Results

Groundwater samples were collected from all permanent monitoring wells installed at Landfill Area 1. These samples were analyzed for VOCs, SVOCs, PAHs (20 percent of samples), pesticides, PCBs, TRPH, and TAL metals. For metals analyses, both unfiltered (total) and filtered (dissolved) groundwater samples were collected.

Organic compounds detected in the permanent monitoring wells were bis(2-ethylhexyl)phthalate, fluoranthene, chlorobenzene, cymene, vinyl chloride, and cis-1,2-dichloroethene (DCE), and TRPH. Of these compounds, only vinyl chloride and chlorobenzene were detected at a concentration exceeding their GCTLs. Vinyl chloride was detected in well LF1DW01 at 2.1 μg/L in July 2001 and at 1.1 μg/L during the Phase II resampling in September 2002. Chlorobenzene was detected in well LF1MW02 at 157 μg/L in July 2001. No PAHs, pesticides, or PCBs were detected in groundwater at the site. Table 4-6 list the organic compounds detected in the monitoring wells. Figure 4-4 illustrates the organic compounds detected in the groundwater at Landfill Area 1.

Total metals concentrations exceeding the GCTLs were detected in all Landfill Area 1 wells. Total iron concentrations greater than the GCTL were observed in all wells. Total iron concentrations ranged from 985 to 23,600 μg/L, which exceed the secondary standard of 300 μg/L. The total and dissolved iron concentrations were comparable. Manganese was another compound detected in all wells, with total concentrations exceeding the secondary standard of 50 μg/L in all wells except LF1W-7-86. Total manganese concentrations ranged from an estimated 17.2J to 643J μg/L. Once again, the total and dissolved manganese results were comparable in each well. In addition, well LF1MW05 contained total aluminum and sodium concentrations that exceeded GCTLs. Only three wells (LF1MW02, LF1MW04, and LF1MW05) contained metals concentrations that exceeded both the respective GCTLs and background concentrations. Analytes detected in these wells that exceeded both screening values included aluminum, iron, manganese, and sodium. Table 4-7 details the total and dissolved (filtered) metals results. Figure 4-5 details total metals concentrations that exceed GCTLs.

4.1.4 Sediment Sample Analytical Results

Five sediment samples (including one duplicate) were collected from the drainage ditch north of Wildwood Road and the creek extending along the eastern side of the site. The samples were analyzed for VOCs, SVOCs, pesticides, PCBs, TRPH, and TAL metals.

Organic compounds were detected in all five samples including Aroclor 1254, Aroclor 1260, 4,4'-dichlorodiphenyldichloroethene (DDE), 2-butanone, acetone, methylene chloride, and TRPH. Aroclor 1260 and 4,4'-DDE were detected above the SQAGs at LF1SD03 with concentrations of 0.151 mg/kg and 0.0096 mg/kg, respectively. Aroclor 1254 was detected at LF1SD04 with a concentration of 0.126 mg/kg, which exceeds the SQAG of 0.00216 mg/kg.

The organic analytical results are summarized in Table 4-8. The distribution of organic compounds detected in sediment is shown on Figure 4-6.

A total of 22 inorganic analytes were detected in the sediment samples at Landfill Area 1. However, only three of the analytes were found exceeding their respective SQAGs. Copper, lead, and mercury were detected at LF1SD03 at concentrations of 151, 111, and 0.45 mg/kg, respectively. These concentrations are also above their respective background concentrations. The only other metals detected above background included calcium and vanadium, neither of which are regulated by the FDEP. Table 4-9 summarizes the metals detected in sediment at Landfill Area 1 and Figure 4-7 displays the distribution of analytes detected above the SQAGs.

4.1.5 Surface Water Analytical Results

Four surface water samples were collected from collocations with the sediment samples. The samples were analyzed for VOCs, SVOCs, pesticides, PCBs, TRPH, total and dissolved TAL metals, alkalinity, and hardness. Alkalinity of the surface water ranged from 46,100 to 263,000 µg/L and hardness ranged from 67,100 to 277,000 µg/L. The only organic constituent detected in surface water was TRPH at an estimated 221J µg/L in LF1SW02. The general chemistry and organic analytical results are summarized in Table 4-10.

Five metals were detected in surface water above their respective surface water cleanup target level (SWCTL) including aluminum, chromium, iron, mercury, and sodium. However, chromium was the only analyte that exceeded both the SWCTL and background concentration. Total chromium was detected in LF1SW02 and LF1SW04 at concentrations of 217 and 30.8 µg/L, respectively. The concentration of dissolved chromium at LF1SW02 was 94.5 µg/L, which also exceeds the SWCTL. A summary of metals detected in surface water is presented as Table 4-11. The distribution of metals detected above SWCTLs is displayed on Figure 4-8.

4.1.6 Landfill Gas Monitoring Results

Gas readings were collected from the surface of Landfill Area 1 and from barhole probes installed near the perimeter. Both an MSA Passport and an FID equipped with a carbon filter were used for the gas monitoring. Table 4-12 provides gas monitoring information for Landfill Area 1. Methane was not detected during the surface emissions monitoring. During the barhole probe monitoring, gas readings were also collected from the piezometers and monitoring wells. Readings were collected from the wells and piezometers for informational purposes only, as the readings are not representative of the vadose soil, due to their screened intervals. No methane was detected from the barhole probes with the Passport instrument. Several barhole probes

contained detectable methane when measured with the FID. Barhole probes with detectable readings included BP-15 and BP-17 through BP-22. The maximum concentration detected was 1.9 parts per million (ppm) from BP-19.

4.2 Landfill Area 3

4.2.1 Surface Soil Analytical Results

Surface soil samples were collected from 20 locations within and around the boundary of Landfill Area 3 during Phase I of the RI. All the samples were analyzed for VOCs, SVOCs, pesticides, PCBs, TRPH, and TAL metals. In addition, 20 percent of the samples were also analyzed for PAHs.

Table 4-13 summarized the organic compounds that were detected in surface soil at Landfill Area 3. Four VOCs (acetone, cymene, methylene chloride and styrene) were detected in surface soil with only methylene chloride reported by the laboratory above its SCTL. However, all reported concentrations were flagged as nondetect (UB or UJ) by data validation. A total of 18 SVOCs were detected in seven samples with only benzo(a)pyrene and dibenz(a,h)anthracene found above the SCTLs. Benzo(a)pyrene was detected in three samples at concentrations ranging from an estimate 0.153J to 0.956 mg/kg. Dibenz(a,h)anthracene was detected at LF3SS11 at an estimated 0.207J mg/kg. Most of the organic compounds found in the surface soil were from samples located in the south-central portion of the site. The distribution of organic compounds detected in soil at Landfill Area 3 is shown in Figure 4-9.

The only PCB detected in surface soil at Landfill Area 3 was Aroclor 1260 at estimated concentrations of 0.0134J and 0.0174 mg/kg in LF3SS08 and LF3SS02, respectively. TRPH was detected in all surface soil samples at concentrations ranging from an estimated 8.38 mg/kg in LF3SB04 to 113 mg/kg in LF3SS05, which is below the SCTL. No pesticides or PAHs were detected in surface soils.

Table 4-14 summarizes metals detected in the surface soil at Landfill Area 3. Twenty-two metals were detected in the surface soil. Of those, 13 were detected above background concentrations with only one, copper, exceeding the SCTL as well as background. Copper was detected in LF3SS03 at 1770 mg/kg.

4.2.2 Subsurface Soil Analytical Results

Subsurface soil samples were collected from eight soil borings located around the perimeter of Landfill Area 3 during Phase I of the RI. The samples were analyzed for the same parameters as the surface soil.

The only organic compounds detected in subsurface soil at Landfill Area 3 included acetone, bis(2-ethylhexly)phthalate, and TRPH, none of which exceeded the SCTLs. Organic analyses results are summarized in Table 4-15 and displayed on Figure 4-9.

Metals results from subsurface soil sampling are summarized in Table 4-16. A total of twenty inorganic elements were reported, none of which exceeded SCTLs. Only two metals, lead and magnesium, were detected above background. Lead was reported in three samples including a duplicate at estimated concentrations ranging from 6J to 7J mg/kg. Magnesium had a concentration of 206J in LF3SB04.

4.2.3 Groundwater Analytical Results

Direct-push and monitoring well groundwater sampling was conducted during Phase I and Phase II of the RI.

4.2.3.1 Direct-Push Groundwater Results

Groundwater samples were collected from two depth intervals at six piezometer locations during Phase I of the RI. Exceptions to this were at LF3PZ01 and LF1PZ04 where refusal occurred at a shallow depth because of flowing sands. All DPT groundwater samples were analyzed for VOCs and SVOCs. Sample AC3014 collected from LF3PZ03 was also analyzed for PAHs. Groundwater samples were collected from three depth intervals at eight DPT locations during the Phase II RI and analyzed for VOCs only. The results of the DPT groundwater sampling at Landfill Area 3 are summarized in Table 4-17.

One DPT groundwater sample collected from the 14 to 18-foot interval at LF3PZ02 contained VOCs exceeding GCTLs. The compounds found above GCTLs included cis-1,2-DCE (17,500 µg/L), trans-1,2-DCE (2,610 µg/L), trichloroethene (TCE) (4,000 µg/L), and vinyl chloride (328 µg/L) during the Phase I investigation. Concentrations of cis-1,2-DCE and TCE fell to estimated 4.4J and 1.2J µg/L, respectively in the 26 to 30-foot interval of the same borehole. The only other VOC detected in groundwater during Phase I was acetone at concentrations ranging from an estimated 38.8J µg/L in the 26 to 30-foot interval of LF3PZ06 to 322J µg/L in the 14 to 18-foot interval of LF3PZ05. Di-n-octyl phthalate and bis(2-hexylethyl)phthalate were the only

SVOCs detected in DPT groundwater samples, neither of which exceeded the GCTL. No PAHs were detected in the sample collected from the 14 to 18-foot interval of LF3PZ03.

The only compounds detected during the Phase II investigation included acetone (6J to 207J µg/L), toluene (0.6J µg/L), m,p-xylene (0.51J µg/L), and tetrachloroethene (PCE) (0.54J µg/L). The detection of PCE was in the deep sample from LF3DP07, which is downgradient of LF3PZ03. The distribution of compounds detected in direct-push groundwater samples is shown of Figure 4-10.

4.2.3.2 Monitoring Well Results

Groundwater samples were collected from seven shallow (5 new and 2 pre-existing) monitoring wells and one deep well during the Phase I investigation and one new shallow and one new deep well during Phase II. The samples were analyzed VOCs, SVOCs, pesticides, PCBs, and TAL metals, with PAH analysis performed on samples from wells LF3DW01, LF3DW02, LF3MW03, and LF3MW06. Tables 4-18 and 4-19 summarize the organic and inorganic analytes detected in monitoring wells, respectively.

Well LF3MW06 was installed during the Phase II fieldwork immediately adjacent to former LF3PZ02 (Figure 4-11) that had been destroyed by logging activities after the completion of the Phase I investigation. Results from this well (samples BC3001 and BC3002 field duplicate) showed the same compounds as the LF3PZ02 results from Phase I and included cis-1,2-DCE (15,700 µg/L), trans-1,2-DCE (2,370 µg/L), TCE (610 µg/L), and vinyl chloride (270 µg/L). In addition, 1,2,4-trimethylbenzene was detected in the original sample (BC3001) at an estimate 193J µg/L, which is also above the GCTL. cis-1,2-DCE was also detected in LF3DW02, which is located downgradient of LF3MW06, at an estimated 29.3J µg/L during the Phase II fieldwork (Figure 4-11). The only VOC detected in monitoring wells during the Phase I investigation was chlorobenzene in LF3W-5-86 at 8.8 µg/L.

The only SVOCs detected in any monitoring well were di-n-octyl phthalate and bis(2-hexylethyl)phthalate in the duplicated sample from LF3W-4-86 (AC3008). The only PAH detected was naphthalene in LF3MW03 at an estimated 1.3J µg/L. The only TRPH detection was that of 1,120 µg/L in LF3W-5-86. No pesticides were detected groundwater at Landfill Area 3.

Twenty-two metals were detected in groundwater at Landfill Area 3 (Table 4-19). Ten analytes (aluminum, barium, iron, lead, magnesium, manganese, sodium [total and dissolved], chromium,

mercury, nickel [total], and potassium [dissolved]) were detected at concentrations exceeding either background or GCTLs. Of these only aluminum, iron, lead, manganese, and mercury were detected above background and the GCTLs. Aluminum concentrations that exceeded background and GCTLs ranged from 2,340 µg/L (dissolved) in LF3W-5-86 to 3,990 µg/L (total) in LF3W-4-86. Iron concentrations above both screening values ranged from 9,190 µg/L (total) in LF3MW05 to 13,000µg/L (dissolved) in LF3W-5-86. Manganese ranged from 177 µg/L (total) in LF3W-5-86 to 1,260 µg/L (dissolve) in LF3MW05. Mercury was detected at 3J µg/L in LF3MW02. With the exception of mercury in LF3MW02, all the excedances occurred in the center and along the southern boundary of the site (Figure 4-12).

4.2.4 Sediment Sample Analytical Results

Four sediment samples were collected from two ponds and a water-filled ditch located within the Landfill Area 3 boundaries. The samples were analyzed for VOCs, SVOCs, pesticides, PCBs, total organic carbon, and TAL metals. Sample AC1005 was also analyzed for PAHs as part of the 20 percent analytical program.

Table 4-20 summarizes the organic compounds detected in sediment at Landfill Area 3. Five VOCs were detected in sediments at concentrations ranging from an estimated 0.0027J mg/kg of toluene at LF3SD01 to 0.474J mg/kg of acetone at LF3SD04. None of the detected VOCs have established SQAGs. TRPH was detected at all four locations at concentrations ranging from 12.8 to 125 mg/kg.

Five SVOCs and nine PAHs were detected in the sediment samples. Those exceeding the SQAGs included benzo(a)anthracene, benzo(a)pyrene, chrysene, fluoranthene, and pyrene, all at LF3SD03. Concentrations ranged from an estimated 0.185 mg/kg of pyrene to 0.528 mg/kg fluoranthene. Aroclor 1260 was detected in four samples (including the duplicate) at concentrations ranging from and estimated 0.0168J mg/kg at LF3SD01 to 0.056J mg/kg at LF3SD03, all of which exceed the SQAG. Figure 4-13 shows the distribution of organics detected in sediment at Landfill Area 3.

Several metals were detected in sediment samples at Landfill Area 3 (Table 4-21). Four metals, arsenic, iron, mercury, and zinc were detected above background concentrations. The detection of mercury in LF3SD03 also exceeded the SQAG with an estimated concentration of 0.22J mg/kg.

4.2.5 Surface Water Analytical Results

Surface water samples were collected from the same locations as the sediment samples and analyzed for VOCs, SVOCs, pesticides, PCBs, TAL metals, alkalinity, and hardness. Samples collected at LF3SW04 were also analyzed for PAHs.

Table 4-22 summarizes the organic compounds detected in surface water as well as the general chemistry of the surface water. The only VOC detected in surface water was cymene in LF3SW03 at 12.9 µg/L. Di-n-octyl phthalate and bis(2-ethylhexyl)phthalate were detected in three samples. Concentrations of bis(2-ethylhexyl)phthalate ranged from an estimated 2.8J to 5.8 µg/L all of which exceed the SWCTL. TRPH was detected in all five samples (including duplicate) at estimated concentrations of 266J to 415J µg/L. Alkalinity ranged from an estimate 4,800J to 21,800J µg/L. Hardness showed good consistency, ranging from 30,500 to 37,500 µg/L.

Several total and dissolved metals were detected in surface water at Landfill Area 3 (Table 4-23). Seventeen analytes had concentrations exceeding either background or SWCTLs. Of these only aluminum, barium, chromium, copper, iron, sodium, and zinc were detected above both screening values. All the analytes exceeding background and SWCTLs were detected in LF3SW03 and ranged from 154 µg/L of copper to 164,000 µg/L of sodium. The distribution of metals exceeding SWCTLs is shown on Figure 4-14.

4.2.6 Landfill Gas Monitoring Results

Gas readings were collected from the surface of Landfill Area 3 and from barhole probes installed near the perimeter. Both an MSA Passport and an FID equipped with a carbon filter were used for the gas monitoring. Table 4-24 provides gas monitoring information for Landfill Area 3. Methane was not detected during the surface emissions monitoring. During the barhole probe monitoring, gas readings were also collected from the piezometers and monitoring wells. Readings were collected from the wells and piezometers for informational purposes only, as the readings are not representative of the vadose soil, due to their screened intervals. No methane was detected from the barhole probes with the Passport instrument. Two barhole probes contained detectable methane when measured with the FID. Barhole probes with detectable readings included BP-20 and BP-23, both of which contained methane readings of 0.1 ppm.

4.3 Comparison of Analytical Results to Background

The following sections summarize the results of comparisons of investigation site soil and groundwater data to background data. The complete site-to-background comparisons for soil,

groundwater, surface water, and sediment are provided in Appendices K (Landfill Area 1) and L (Landfill Area 3).

4.3.1 Comparison of Site and Background Soil Data

The Landfill Areas 1 and 3 site soil data sets were subjected to a combined statistical and geochemical evaluation to determine if the metals concentrations detected in soil are naturally occurring (Appendices K-1 and L-1). Multiple statistical tests were performed in parallel as an initial screening step, to test for different modes of contamination. The hot measurement test, in which the site maximum detected concentration is compared to the background screening value (the 95th upper tolerance limit [UTL_{95}] or 95th upper confidence limit of the 95th percentile [$UCLP_{95}$] of the background distribution), was performed to detect potential hot spots. The Wilcoxon rank sum (WRS) test, which tests for differences in the medians of two data sets, was performed because of its sensitivity in detecting slight but pervasive contamination. Box-and-whisker plots were prepared to visually compare the site and background data sets and to properly interpret the results of the WRS test. Any analyte that failed one or both statistical tests was then subjected to geochemical evaluation, which is based on the natural associations of trace elements with specific minerals in the soil matrix. Correlation plots of trace elements versus major elements were constructed to explore these associations.

Landfill Area 1. Twenty of the twenty-three TAL metals (all except aluminum, selenium, and thallium) failed statistical comparison to background and were subjected to geochemical evaluation (Appendix K-1). For most of these elements, the elevated concentrations could be explained as most likely resulting from the preferential enrichment of samples with minerals such as iron oxides, manganese oxides, and clays, which naturally concentrate specific trace elements. Surface soil sample AA0030 and subsurface soil sample AA0031 exhibit anomalous concentrations of antimony that may contain a component of contamination.

Landfill Area 3. Barium, beryllium, cadmium, calcium, chromium, copper, iron, lead, magnesium, manganese, mercury, nickel, potassium, silver, sodium, and zinc were subjected to geochemical evaluation (Appendix L-1). For most of these elements, the elevated concentrations could be explained as most likely resulting from the preferential enrichment of samples with minerals such as iron oxides and manganese oxides, which naturally concentrate specific trace elements. However, cadmium, copper, lead, mercury, nickel, silver, and zinc concentrations were anomalously high in at least one sample each, and those anomalous concentrations may contain a component of contamination (Appendix L-1).

4.3.2 Comparison of Site and Background Groundwater Data

The groundwater data sets for Landfill Areas 1 and 3 were subjected to a combined statistical and geochemical evaluation to determine if the metals concentrations detected in groundwater are naturally occurring (Appendices K-2 and L-2). The same statistical procedures described in Section 4.3.1 were performed for the groundwater evaluation, and any analyte that failed one or both statistical tests was subjected to geochemical evaluation.

The primary mechanisms examined in the groundwater geochemical evaluation were the effects of suspended particulates and reductive dissolution. In neutral pH, moderate to oxidizing redox conditions, detectable aluminum and iron concentrations above about 1 milligrams per liter (mg/L) indicate the presence of suspended clay minerals and iron oxide minerals, respectively. The presence of trace elements adsorbed onto suspended particulates can greatly increase trace element concentrations as reported by an analytical laboratory. These adsorbed trace elements are not in true solution, and can be removed by settling or filtration. The same concepts involved in the evaluation of soil data also apply to groundwater data: samples containing trace elements adsorbed on suspended clay particulates (which contain aluminum) should show a positive correlation with aluminum concentrations, and samples containing trace elements adsorbed on suspended iron oxides should show a positive correlation with iron concentrations. Under reducing conditions, the solubilities of iron oxide and manganese oxide minerals increases, and thus the dissolved concentrations of iron and manganese will increase, as will the dissolved concentrations of associated trace elements. These reducing conditions can be natural, as in swamps and wetlands, or they can be caused by the anaerobic degradation of organic contaminants such as fuels and chlorinated solvents. Correlation plots of trace elements versus major elements were constructed to explore these associations.

Landfill Area 1. Aluminum, arsenic, calcium, chromium, iron, lead, magnesium, manganese, potassium, sodium, and zinc were subjected to geochemical evaluation (Appendix K-2). Elevated concentrations of most of these elements are due to the presence of suspended particulates, naturally-occurring major cations, or naturally-reducing conditions. Arsenic concentrations in samples AA3006 and AA3024, iron concentrations in samples AA3006 and AA3024, and the manganese concentration in sample AA3006 are most likely elevated due to reductive dissolution, as a secondary effect of the organic contamination at those locations. The zinc concentration in sample AA3006 is anomalously high and there may be a component of contamination in this sample.

Landfill Area 3. Aluminum, barium, chromium, iron, lead, magnesium, manganese, mercury, nickel, potassium, and sodium were subjected to geochemical evaluation (Appendix L-2). Elevated concentrations of aluminum, lead, and mercury are associated with suspended particulates, and are natural. Magnesium, potassium, and sodium are major cations that are primary components of groundwater, and are natural. Concentrations of barium in samples AC3009 and AC3024; chromium in samples AC3001 and AC3009; iron in sample AC3009; nickel in AC3009; and manganese in sample AC3024 are anomalously high and may reflect contamination. It is important to note that the iron concentration in sample AC3009 may be elevated due to reductive dissolution, as a secondary effect of the organic contamination at that location.

4.3.3 Comparison of Site and Background Sediment Data

The Landfill Areas 1 and 3 sediment data sets were subjected to a combined statistical and geochemical evaluation to determine if the metals concentrations detected in groundwater are naturally occurring (Appendices K-3 and L-3). The same statistical procedures described in Section 4.3.1 were performed for the sediment evaluation, and any analyte that failed one or both statistical tests was subjected to geochemical evaluation, as described for soil in Section 4.3.1.

Landfill Area 1. Aluminum, calcium, copper, lead, magnesium, manganese, mercury, vanadium, and zinc were subjected to geochemical evaluation (Appendix K-3). Elevated concentrations of these elements most likely result from the preferential enrichment of samples with organic carbon or minerals such as iron oxides and clays, which naturally concentrate specific trace elements. Inorganic contamination is not indicated in the Landfill Area 1 sediment samples.

Landfill Area 3. Aluminum, arsenic, iron, mercury, and zinc were subjected to geochemical evaluation (Appendix L-3). Elevated concentrations of aluminum, mercury, and zinc most likely result from the preferential enrichment of samples with organic carbon or minerals such as manganese oxides and clays, which naturally concentrate specific trace elements. Sample AC1006 contains anomalously high arsenic and iron concentrations that may contain a component of contamination.

4.3.4 Comparison of Site and Background Surface Water Data

The Landfill Areas 1 and 3 surface water data sets were subjected to a combined statistical and geochemical evaluation to determine if the metals concentrations detected in groundwater are naturally occurring (Appendices K-4 and L-4). The same statistical procedures described in

Section 4.3.1 were performed for the surface water evaluation, and any analyte that failed one or both statistical tests was subjected to geochemical evaluation, as described for groundwater in Section 4.3.2.

Landfill Area 1. Antimony, calcium, chromium, lead, nickel, and thallium were subjected to geochemical evaluation (Appendix K-4). Elevated concentrations of antimony, chromium, lead, nickel, and thallium are most likely due to the presence of suspended particulates, and are natural. Calcium is a naturally-occurring major cation and its concentrations in the site samples are also natural. Inorganic contamination is not indicated in the Landfill Area 1 surface water samples.

Landfill Area 3. Chromium, iron, lead, nickel, and potassium were subjected to geochemical evaluation (Appendix L-4). Elevated concentrations of these elements are most likely due to the presence of suspended particulates or naturally-reducing conditions, and are natural. Inorganic contamination is not indicated in the Landfill Area 3 surface water samples.

5.0 Baseline Human Health Risk Assessment

This chapter presents the conceptual site exposure model (CSEM), the protocol, and the results of the baseline human health risk assessment (BHHRA) for Landfill Areas 1 and 3 at the LFNAS.

5.1 Conceptual Site Exposure Model

The CSEM provides the basis for identifying and evaluating the potential risk to human health (EPA, 1989a). The CSEM includes all plausible receptors for all potential site-use scenarios. Graphically presenting all possible pathways by which a potential receptor may be exposed, including all sources, release and transport pathways, and exposure routes facilitates consistent and comprehensive evaluation of risk to human health, and helps to ensure that potential pathways are not overlooked. The elements necessary to construct a complete exposure pathway and develop the CSEM include:

- Physical setting
- Source of contamination potentially released at site
- Source media (e.g., soil, groundwater)
- Contaminant release mechanisms
- Contaminant transport pathways
- Transport (secondary, tertiary) media
- Exposure media (e.g., soil, drinking water, air)
- Receptors
- Routes of exposure (ingestion, inhalation, dermal uptake).

Contaminant release mechanisms and transport pathways, which result in the transfer of contaminants across media, are not relevant for direct receptor contact with a contaminated source medium. Figure 5-1 presents the human health CSEM for Landfill Areas 1 and 3. Both landfills are identical regarding the elements bulleted above.

5.1.1 Physical Setting

The LFNAS has three former landfill areas designated Landfill Areas 1, 2, and 3, which were utilized for disposal from the 1940s until the early 1970s. Landfill Area 2 was previously evaluated and is not considered herein. The types of materials disposed in the former landfills are unknown. It is also unknown whether trenches were utilized at the former landfills. It appears that the landfills were not capped but were covered with soil. The former landfills are within a swampy, marshy area of poorly drained soil, which suggests that the top of the shallow

unconfined aquifer is very close to the surface. It should be noted that only the shallow unconfined aquifer is evaluated herein.

Although the former landfills are not currently used, residential and industrial areas are nearby, as is a golf course and wildlife refuge. Both former landfills are completely within the limits of the Reynolds Industrial Park; however, Landfill Area 1 is also adjacent to a wildlife refuge, but Landfill Area 3 is surrounded by industrial businesses.

Landfill Area 1. Landfill Area 1 is a large 6.2-acre field somewhat higher than the existing terrain (IT, 2000b). It is covered with high brush, although some areas of bare soil are present. A small creek or ditch along the eastern edge contains water; however, it is too small to support fish that would be caught for consumption; across the creek is the wildlife refuge property. Wildwood Road runs along the northern edge of the landfill; a large ditch containing water exists just north of the road. A wetland forest lies just south of the landfill. The northern side of the landfill is fenced; however, access to the former landfill is easily gained. Landfill Area 1 provides acceptable habitat for game animals such as deer and wild turkey.

Landfill Area 3. Landfill Area 3 is a heavily wooded 7.4-acre area with pines with an average breast diameter of approximately 8 inches (IT, 2000b). The area around Landfill Area 3 consists of a similar pine forest. A small shallow pond exists within the landfill proper. It is too small to support fish that would be caught for consumption. Landfill Area 3 provides acceptable habitat for game animals such as deer and wild turkey.

5.1.2 Sources of Contamination Potentially Released at Site

The source of the contamination potentially released at the former landfills is the waste materials dumped or buried at the site. In addition, it is possible that waste materials could have been spilled or leaked near the landfills en route to their final disposal site.

5.1.3 Media of Interest

Potentially contaminated source media are probably limited to surface and subsurface soil. Since the top of the shallow unconfined aquifer is near the surface, it is possible that waste may have been disposed directly into shallow groundwater. Potential exposure media relevant to the LFNAS former landfills include:

- Groundwater (shallow unconfined aquifer only)
- Surface soil
- Subsurface soil

- Surface water
- Sediment
- Ambient air.

Identification of source media is most important when all media of concern were not or cannot be sampled, and modeling is required to estimate chemical concentrations. In this case, all media of interest except ambient air were adequately sampled. Surface soil samples were taken from 0 to 1 foot bgs; subsurface soil samples were taken from 1 to 3 feet bgs. A medium called "total soil" was created by combining the surface soil data and subsurface soil data to reflect the potential for future development to involve excavation, earth-moving or grading.

Surface water and sediment samples for Landfill Area 1 were taken from outside the landfill proper. Surface water and sediment samples for Landfill Area 3 were taken from within the landfill proper.

5.1.4 Release Mechanisms and Transport Pathways

Potential contaminant release mechanisms and transport pathways relevant to risk assessment of the former landfills include the following:

- Release of soil dust and volatilization of VOCs from soil or groundwater into air.
- Runoff and erosion to surface water and sediment.
- Leaching of chemicals for soil to groundwater.
- Groundwater discharge to surface water.

5.1.5 Receptors and Exposure Pathways

Several receptors may be exposed to the various media at the LFNAS former landfills under both current and future site-use assumptions. The receptor scenarios are developed in detail for Landfill Area 1, and variations on these receptor scenarios are devised to fit the unique features of Landfill Area 3.

5.1.5.1 Landfill Area 1

Currently, Landfill Area 1 is not used; however, the northern edge of the landfill along Wildwood Road is maintained, while the eastern and southern portions of the landfill adjoin a wildlife refuge. No future site use is planned for Landfill Area 1; however, if the site were developed, the most likely future site use would be industrial or commercial. It is unlikely that the area of the landfill proper would be used as a building site, but its use as a grassy mall, landscaped garden area or a parking lot is reasonable. The surrounding areas, within Reynolds Industrial Park, may be used for building. It is also possible, although unlikely, that the area

might be developed for residential use. Homes probably would not be placed on the area of the landfill proper, but they could be placed on the surrounding areas, giving residents access to the area of the landfill proper. In summary, 'current site use' refers to the present undeveloped state of the former landfill; 'future site use' refers to further development for any of the purposes mentioned above. Plausible receptors under the current site-use scenario include a groundskeeper and a trespasser. Plausible receptors under the future site-use scenario include groundskeepers, trespassers or site visitors, construction workers, and residents.

Surface and subsurface soil samples were taken within the area of the landfill proper as well as from the surrounding area. Potential exposure to soil for the receptors mentioned above under both current and future site use are summarized as follows:

Groundskeeper	
Current site use	Surface soil outside the landfill proper
Future site use	Total soil both inside and outside the landfill proper (to cover the possibility that future development may involve excavation that would bring subsurface soil to the surface).
Construction Worker	
Current site use	No plausible exposure to soil
Future site use	Total soil both inside and outside the landfill proper
Trespasser (called Site Visitor under future site use)	
Current site use	Surface soil both inside and outside landfill proper
Future site use	Total soil both inside and outside the landfill proper
Resident	
Current site use	No plausible exposure to soil
Future site use	Total soil both inside and outside the landfill proper

5.1.5.1.1 Groundskeeper

Site workers are plausible receptors under both the current and future site-use scenarios. Various site workers may be exposed to Landfill Area 1 and the area near the landfill (if it is developed), including maintenance workers, groundskeepers, office workers, delivery personnel, etc. A groundskeeper would represent the upper bound for exposure for all site workers, because he may be involved in landscaping, gardening, lawn maintenance and other activities resulting in maximum contact with soil. Therefore, the groundskeeper is the only site worker evaluated.

Potential exposure pathways under the current site use scenario include incidental ingestion of surface soil, dermal contact, and inhalation of airborne dust and VOCs that volatilize from the surface soil. Potential exposure pathways under the future site use scenario are the same but involve total soil rather than just surface soil, assuming that some development or grading may have occurred to enable exposure to subsurface soil.

Inhalation of VOC emissions from shallow unconfined groundwater is another potentially complete pathway. However, the presence of overlying soil, natural air currents, and the large dilution factor of ambient air generally reduces airborne concentrations from groundwater to toxicologically insignificant levels. Furthermore, the exposure from shallow groundwater developed as a source of potable water, which would be much greater, is evaluated as described in the next paragraph.

It is assumed that in the future the shallow unconfined aquifer is developed as a source of potable water. The groundskeeper ingests the groundwater as a source of drinking water, and he experiences dermal contact from using the groundwater to water lawns or clean equipment. Inhalation of VOCs from groundwater during its use is a potentially complete pathway, but natural air currents and the large dilution factor of ambient air are expected to reduce airborne concentrations to toxicologically insignificant levels, especially relative to drinking water consumption and dermal contact, both of which are evaluated.

Exposure to surface water and sediment in the ditch north of Wildwood Road is possible; however, this is likely to be accidental rather than as part of the groundskeeper's regular duties, and is more appropriately classified as acute rather than chronic exposure. Since exposure to these media is not predictable and since acute exposures are not evaluated in the same way as chronic exposures, exposure to surface water and sediment is not quantified for the groundskeeper.

5.1.5.1.2 Construction Worker

Currently, there is no construction occurring at the site, nor is any anticipated in the near future. Therefore, the construction worker is evaluated only under the future site-use scenario. The construction worker is introduced as a plausible receptor for short-term exposure to total soil (surface and subsurface soil combined) and to shallow unconfined groundwater developed as a source of potable water under the future site-use scenario. The construction worker may be involved in a hypothetical 6-month construction project that includes excavation and grading of soil. The soil exposure pathways described for the groundskeeper also apply to the construction worker. In addition, the construction worker is evaluated for dermal exposure to shallow unconfined groundwater, and to inhalation of VOC emissions from groundwater, because groundwater levels at both landfills frequently occur within 3 feet bgs (Appendix M).

It is assumed that in the future the shallow unconfined aquifer is developed as a source of potable water. The construction worker ingests the groundwater as a source of drinking water, and he

experiences dermal contact from using the groundwater to clean equipment or rinse perspiration from his body. Inhalation of VOCs from groundwater during its use as potable water is a potentially complete pathway, but natural air currents and the large dilution factor of ambient air are expected to reduce airborne concentrations to toxicologically insignificant levels.

Exposure to surface water and sediment in the ditch north of Wildwood Road is possible; however, this is likely to be accidental rather than as part of the construction worker's regular duties, and is more appropriately classified as acute rather than chronic exposure. Since exposure to these media is not predictable and since acute exposures are not evaluated in the same way as chronic exposures, exposure to surface water and sediment is not quantified for the construction worker.

5.1.5.1.3 Trespasser or Site Visitor

The trespasser is a plausible receptor under the current site-use scenario. The trespasser may walk, hike, or play games on site, and thereby become exposed to soil. Under the current site-use scenario, the trespasser would be exposed only to surface soil by the same pathways as the groundskeeper and construction worker. The trespasser may also be a plausible receptor under the future site-use scenario, in which case he is more appropriately called a site visitor and would be exposed to total soil.

The trespasser may also wade in the ditch located north of Wildwood Road, thereby becoming exposed directly to surface water and sediment. Plausible exposure pathways include incidental ingestion and dermal contact. Inhalation of airborne VOCs from surface water is a potentially complete pathway, but natural air currents and the large volume of ambient air are expected to reduce concentrations to toxicologically insignificant levels.

5.1.5.1.4 Resident

Currently, there are no residents at the site or within the Reynolds Industrial Park; therefore, the resident is not evaluated under the current site-use scenario. Since the site is within an industrial park, the resident is not a likely receptor under the future site-use scenario; however, EPA Region 4 guidelines (EPA, 2002a) suggest that a residential scenario be evaluated in order to provide a more complete risk assessment and more information to risk managers. Exposure is evaluated by assuming the resident spends part of his time as a child and as an adult.

Potential future exposure pathways for the resident include incidental ingestion, dermal contact, inhalation of airborne dust, and inhalation of airborne VOCs from total soil. It is assumed that in

the future the shallow unconfined aquifer is developed as a source of potable water. Plausible exposure routes include ingestion of drinking water, dermal contact, and inhalation of airborne VOC emissions during household use.

Also, the resident may wade in the ditch located north of Wildwood Road, thereby becoming exposed directly to surface water and sediment. Plausible exposure pathways include incidental ingestion and dermal contact. Inhalation of VOCs from surface water is expected to be toxicologically insignificant compared with incidental ingestion and dermal contact as explained for the trespasser.

5.1.5.1.5 Receptor Scenarios Not Evaluated

Currently, Landfill Area 1 provides acceptable habitat for deer, wild turkey, and other game animals. Hunting, however, is illegal within Green Cove Springs and the hunter would need to trespass in order to hunt at the site. Although such a scenario is possible, the type of illegal hunting likely to occur on landfills surrounded by an industrial park would be limited to varmint shooting. The harvesting of game for human consumption is unlikely. Therefore, indirect exposure to soil by consumption of game animals is not evaluated. As noted above, a ditch containing water is located north of Wildwood Road. The ditch is insufficient to support fish that are likely to be caught and consumed by an angler. Therefore, indirect exposure to surface water or sediment by evaluation of fish consumption is not evaluated.

5.1.5.2 Landfill Area 3

The receptor scenarios developed above for Landfill Area 1, including the evaluation of exposure to soil inside and outside the landfill boundaries, apply fully to Landfill Area 3. Surface water is present at Landfill Area 3 in the form of a small shallow pond within the landfill proper. Therefore, the wading scenario resulting in exposure to surface water and sediment developed for the trespasser or site visitor and the resident for Landfill Area 1 also apply to Landfill Area 3.

5.2 Risk Assessment Tasks

The human health risk assessment for Landfill Areas 1 and 3 at LFNAS, Clay County, Florida, was performed in two phases. Phase I is a screening procedure that consists of the following three steps.

- The first step is to compile a list of all the chemicals with concentrations above method detection limits (MDL) in all of the media sampled.

- The second step is to restrict the list to site-related chemicals by removing those that are present at background concentrations.

- The third step reduces the list further to identify chemicals of potential concern (COPC), which are the site-related chemicals that may contribute significantly to risk.

COPCs are carried forward to Phase Two, the site-specific BHHRA.

5.2.1 Phase I: Screening Human Health Risk Assessment

Phase I, or the screening human health risk assessment, consists of several steps that determine if COPCs are present at the landfills. The first step creates a list of all chemicals detected at least once in a given medium.

All analytical data were validated. Some of the more common validation qualifier codes and their meanings are (EPA, 1989a):

- U: Chemical was analyzed for but not detected; the associated value is the reporting limit (RL).

- J: Value is estimated, between the MDL and the RL.

- R: Quality control indicates that the data are unusable (chemical may or may not be present).

- B: Concentration of chemical in sample is not sufficiently higher than concentration in the blank (using 5-times, 10-times rule, as follows).

Organic chemicals are omitted from consideration if they are common laboratory contaminants (acetone, 2-butanone, methylene chloride, toluene, phthalate esters) and if all sample concentrations are less than 10 times the highest blank concentration. Other organic chemicals are omitted if all analytical results are less than 5 times the highest concentration detected in any blank.

"J" qualified data are used in the risk assessment; "R" and "B" qualified data are not. "U" qualified data for a given chemical in a given medium are used in the risk assessment, provided that the chemical was detected in one or more samples. One-half the RL is used as a surrogate concentration for the "U" qualified data point in all statistical analyses.

5.2.1.1 Selecting Chemicals of Potential Concern

The following approach was used to select COPCs. The process of elimination proceeds in the order of the criteria listed below.

5.2.1.2 Background Screening

The first step in the human health risk assessment screening is to compare the site-specific maximum detected concentrations (MDC) of inorganics and PAHs with their respective background screening criteria (BSC) at LFNAS, determined during the background study. The background studies and background screening are discussed in Chapter 4.0 of this report. Other lines of evidence were also used to identify chemicals present at background concentrations. Inorganics or PAHs determined not to be site-related because of the background study were screened out and not identified as COPCs. Site-related chemicals are carried to the next step.

5.2.1.3 Exclusion of Essential Elements

Major nutritionally essential elements (calcium, sodium, potassium, and magnesium) are excluded from consideration in the BHHRA. The nutritional element iron is carried through the COPC selection process; however, due to concerns with the oral reference dose (RfD) for iron, it is not included in the quantitative risk assessment. Instead, it is discussed in the toxicity assessment (Section 5.6).

5.2.1.4 Risk-Based Screening

This step of the human health risk assessment screening consists of comparing the MDCs with risk-based screening levels, based upon the FDEP residential soil and groundwater cleanup target levels (CTL) (FDEP, 1999). Noncancer CTLs are divided by a factor of 10 to reflect a hazard index (HI) of 0.1. CTLs based on organoleptic or nuisance considerations are not reduced, because they currently reflect cancer risk levels less than 1E-6 or HI values less than 1. CTLs based on detection limits are not reduced because lower values would be unenforceable.

Soil CTLs are not presented for inorganic lead. To account for total lead in soil, a screening value of 400 mg/kg will be utilized, as per EPA (1998). To account for total lead in groundwater, the national action level for lead in tap water of 0.015 mg/L (EPA, 2002b) is used in place of a CTL.

Chemicals with MDCs less than their respective CTLs are not considered further in the risk assessment because it is very unlikely that they would contribute significant risk. If no site-related chemical in a medium exceeds its CTL, then that medium, and its exposure pathways, are

eliminated from the risk assessment process. If the MDC exceeds background and its CTL, the chemical is selected as a COPC for that medium and is carried to Phase II.

5.2.1.5 Chemicals Not Detected

It is not uncommon to have several chemicals in a given medium for which all analytical results are reported as "not detected." This is no cause for alarm when their MDLs are below their CTLs, because it is unlikely that the chemicals, if present at their MDLs, would contribute significantly to site risk. It is possible, however, that undetected chemicals with CTLs below their MDLs could be present at concentrations associated with significant risk. Such chemicals are addressed qualitatively in the uncertainty analysis section.

5.2.2 Baseline Human Health Risk Assessment

This section presents the methodology and assumptions that are used to conduct Phase II, a human health BHHRA for Landfill Areas 1 and 3 at LFNAS. The following approach is based on federal EPA and EPA Region 4 (EPA, 2002a) guidance.

A human health BHHRA consists of the following:

- ***Data Evaluation.*** Describes the statistical summarization of the analytical data, selection of COPCs for each site and each medium of interest (accomplished in Phase I), and the estimation of source-term concentrations (STC) for each COPC in each medium.

- ***Exposure Assessment.*** Further develops the exposure scenarios identified in the CSEM and the rationale by which plausible receptors are selected, the pathways by which they may be exposed, the exposure-point concentrations (EPC) of COPCs, and the estimated dose or dermal uptake rates for each of the COPCs.

- ***Toxicity Assessment.*** Describes the hazard evaluation, i.e., the adverse health effects associated with each of the COPCs, and the dose-response evaluation, i.e., the relationship between dose rate and the magnitude of the adverse effect.

- ***Risk Characterization.*** Combines the output of the exposure assessment and the toxicity assessment to quantify the risk to each receptor.

- ***Development of Remedial Goal Options.*** Selects chemicals of concern (COC) that drive the risk assessment, and estimates remedial goal options (RGO) for the chemicals and media of concern, including both applicable or relevant and appropriate requirements (ARAR) and human health-based cleanup goals for all media considered.

- ***Uncertainty Analysis.*** Describes the sources of uncertainty in the risk assessment and discusses their impact on the results and their interpretation.

5.2.2.1 Data Evaluation

During Phase I, COPCs were determined for the various media at the LFNAS Landfill Areas 1 and 3. The analytical results for the COPCs are subjected to a statistical summarization that permits the calculation of STCs for chemicals in all media that are conservative (high-end) estimates of average. Groundwater is treated as a special case (see below). The process of statistical summarization and development of the STCs is described below.

Once the data set is complete, statistical methods were used to evaluate the analytical results from the sampling program in order to establish the STCs (EPA, 1989a). The statistical methods used in data evaluation reflect EPA guidance. The following information is tabulated for each detected chemical in each medium:

- Chemical name
- Frequency of detection
- Percentage of hits
- Range of detected concentrations
- Range of RLs
- Statistical distribution
- Arithmetic mean
- 95 Percent upper confidence limit (UCL) on the mean of the concentration (if appropriate)
- BSC, which is the 95 percent UTL on the mean of the background concentration or the 95th percentile (for background screening)
- Appropriate CTL
- COPC selection
- STC.

Because of the uncertainty associated with characterizing contamination in environmental media, both the arithmetic mean and the UCL on the mean for either a normal, lognormal, or nonparametric distribution are reported for each COPC in each medium of interest provided the

data are sufficient to characterize the nature of the data distribution with reasonable confidence (i.e., there are at least 5 sample points).

The 95 percent upper confidence limit on the mean is generally referred to as the UCL. In general, outliers are included in the calculation of the UCL because high values seldom appear as outliers for a lognormal distribution, and environmental data are frequently lognormally distributed (EPA, 1992a). Inclusion of outliers increases the overall conservatism of the risk estimate.

The first step in UCL estimation is to determine the statistical distribution of the data set. Data sets with 5 or more samples are tested for normality and lognormality by the Shapiro-Wilks test (EPA, 1992b). The UCL is calculated for the type of distribution (normal or lognormal) that gives the better fit.

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

$$UCL = \bar{x} + t_{1-\alpha,n-1} \bullet (s/\sqrt{n}) \qquad \text{Eq. 5-1}$$

where:

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

The UCL is calculated for data sets identified as lognormal and those determined to fit neither a normal nor a lognormal distribution (nonparametric) by an adaptation of the Chebychev theorem as developed by (EPA, 1997a). The Chebychev equation for calculating the UCL on the mean is as follows:

$$\text{UCL} = \bar{x} + ks/n^{0.5} \qquad \text{Eq. 5-2}$$

where:

$\bar{x}$	=	sample arithmetic mean
k	=	4.47 (unitless)
s	=	sample standard deviation
n	=	number of samples in the data set.

For all media except groundwater, the UCL or the MDC, whichever is lower, is selected as the STC (EPA, 1992a), which is interpreted as a conservative estimate of average. The STC for all groundwater COPC is the MDC.

Analytical results are presented as "nondetects" whenever chemical concentrations in samples do not exceed the MDLs for the analytical procedures for those samples. Generally, the RL is the lowest concentration of a chemical that can be quantified reliably; however, "J" qualified data (those estimated to be between the MDL and the RL) are used in the risk assessment. To apply the previously mentioned statistical procedures to a data set with nondetects, a concentration value must be assigned to the nondetects. Nondetects are assumed to be present at one-half the reporting limit (EPA, 1989a).

5.2.2.2 Data Evaluation for Groundwater

Potential groundwater contamination may be evaluated under the following different scenarios.

Current Groundwater Conditions. The current condition of the groundwater is evaluated by using the analytical results from groundwater samples. MDCs are used to select the groundwater COPC in the Phase I screening described above, and are also adopted as the STCs.

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

- Releases to the soil are old, depth to groundwater is shallow, contaminants released are highly mobile in soil (e.g., VOCs); therefore, it is likely that current groundwater concentrations reflect the full potential for leaching from subsurface soil.

- Contaminants released to soil exhibit very low mobility in the environment, and, in the absence of the release of other solvents, are not expected to migrate to groundwater.

- Geological features render migration to groundwater unlikely.

- Geological features provide little impediment to migration to groundwater, and it is likely that the full potential for leaching is captured in the current groundwater data.

- Repeated sampling reveals concentrations in groundwater holding steady or decreasing, but not increasing with time.

Reasons to decide *for* evaluating future groundwater conditions include:

- Releases to the soil are recent or depth to groundwater is great; it is likely that the full potential for leaching has yet to occur.

- Contaminants released to soil exhibit relatively high mobility in the environment, and are expected to migrate to groundwater.

- Geological features are likely to delay but not prevent significant migration to groundwater.

- Repeated sampling reveals concentrations in groundwater increasing with time.

Groundwater at the LFNAS Landfill Areas 1 and 3 is shallow, suggesting that there is little impediment to leaching to groundwater. Also, the wide range of chemicals detected in groundwater suggests that maximum leaching may have already occurred. Therefore, it is judged that the current groundwater evaluation captures the extent of potential groundwater contamination and that evaluation of future groundwater conditions is unnecessary.

5.3 Exposure Assessment

Exposure is the contact of a receptor with a chemical or physical agent. An exposure assessment estimates the type and magnitude of potential exposure of receptors to COPCs found at a site (EPA, 1989a). An exposure assessment includes the following steps:

- Characterize the physical setting
- Identify the contaminant sources, release mechanisms, and migration pathways
- Identify the potentially exposed populations or receptors
- Identify the potential exposure pathways
- Estimate exposure concentrations
- Estimate chemical intake or uptake rates.

Several of these steps, through identifying the potentially exposed receptors and exposure pathways, were described in Section 5.1. This section more fully develops the receptor scenarios identified in the CSEM. It also develops the protocol for estimating EPCs and for estimating chemical intake or uptake rates.

5.3.1 Receptor Scenarios

The receptor scenarios identified in Section 5.1 are more fully developed below. The exposure pathways are summarized in Table M-1; the exposure variable values are summarized in Table M-2.

5.3.1.1 Groundskeeper

The groundskeeper is identified as a plausible current receptor exposed to surface soil by incidental ingestion, dermal contact, inhalation of airborne dust and inhalation of airborne VOCs, and a plausible future receptor exposed to total soil (through the same soil exposure pathways) and to shallow groundwater developed as a source of drinking water. It is assumed that the groundskeeper would be exposed to groundwater through consumption of drinking water and dermal contact.

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

Clothing provides partial protection against dermal contact with soil, restricting potential contact to approximately 25 percent of the body, or approximately 5,000 square centimeters [cm^2]) (EPA, 1992c). EPA (1992c) recommends a default value of 0.2 mg per square centimeter (mg/cm^2), the lower end of the range of 0.2 to 1.0 mg/cm^2, as an average coefficient for soil-to-skin adherence.

In the future site-use scenario, the groundskeeper may be exposed to shallow unconfined groundwater, which could be developed as a source of drinking water. His drinking water ingestion rate is assumed to be 1 liter per day (L/day) (EPA, 1991). He may also experience dermal contact with groundwater used for irrigation, to clean equipment and to rinse dust or perspiration from his body. For this evaluation, it is assumed that the head, arms, and hands, approximately 4,100 cm^2 (EPA, 1992c), are exposed intermittently throughout the day for up to an hour per day. Because exposure is intermittent, rather than continuous, it is assumed that organic chemical uptake across the dermis does not reach steady state, which guides selection of the EPA (1992c) model used to quantify this pathway.

5.3.1.2 Construction Worker

The construction worker is identified as a plausible future receptor at Landfill Areas 1 and 3. Construction projects are assumed to last for 6 months. The construction worker is assumed to be exposed to total soil (surface and subsurface soil) through incidental ingestion, dermal contact, inhalation of dust, and inhalation of volatilizing VOCs.

He is also assumed to be exposed to shallow unconfined groundwater pooling in shallow trenches or pits by dermal contact and inhalation of airborne VOCs. Exposure to shallow unconfined groundwater developed as a source of drinking water (by ingestion of drinking water and dermal contact) is also assumed under the future land-use scenario.

The construction worker is assumed to be a 70-kg adult who works 8 hours per day, approximately 5 days per week for 6 months of the year on site for a total of 125 days (250 days per year times 0.5 years) (EPA, 1991). The respiratory rate for the construction worker is assumed to be 20 m^3/8-hour workday (2.5 m^3/hour). The soil incidental ingestion rate is assumed to be 100 milligrams per day (mg/day), comparable to an agricultural worker. Clothing provides partial protection against dermal contact with soil, restricting potential contact to approximately 25 percent of the body, or approximately 5,000 cm^2 (EPA, 1992c). EPA (1992c) recommends a default value of 0.2 mg/cm^2, the lower end of the range of 0.2 to 1.0 mg/cm^2, as an average coefficient for soil-to-skin adherence.

Certain assumptions are adopted to evaluate exposure to shallow groundwater (not developed as a potable source) pooling in trenches during construction. It is assumed that the construction period involves work in a utilities trench that is 2 feet (51 cm) wide, 2.5 feet (76 cm) deep, and of an undetermined length. It is assumed that the construction worker wears protective footwear, so that only his hands and forearms are exposed to groundwater in the trench. These body regions constitute approximately 2,000 cm^2 (EPA, 1992c). Dermal exposure to groundwater is assumed to occur intermittently up to 4 hours per day. Because exposure is intermittent, rather than continuous, it is assumed that organic chemical uptake across the dermis does not reach steady state, which guides selection of the EPA (1992c) model used to quantify this pathway.

The construction worker is assumed to be exposed to also groundwater developed as a source of drinking water. His drinking water ingestion rate is assumed to be 1 L/day (EPA, 1991). He may also experience dermal contact with drinking water used to clean equipment and to rinse dust or perspiration from his body. For this evaluation it is assumed that the head, arms and hands, approximately 4,100 cm^2 (EPA, 1992c), are exposed intermittently throughout the day for

up to an hour per day. Because exposure is intermittent, rather than continuous, it is assumed that organic chemical uptake across the dermis does not reach steady state, which guides selection of the EPA (1992c) model to be used to quantify this pathway.

5.3.1.3 Trespasser

The trespasser (or site visitor) is identified as a plausible current and future receptor exposed to soil by incidental ingestion, dermal contact, inhalation of airborne dust, and inhalation of airborne VOCs. The trespasser is assumed to make regular visits to the landfill for recreational purposes. It is assumed that the trespasser makes one visit per week (52 days per year), and spends 6 hours per day on site. The 6 hours per day is assumed to be split so that 4 hours per day are spent in contact with soil and 2 hours per day are spent wading in surface water. Exposure to sediment parallels exposure to surface water.

EPA (2002a) defines the trespasser as a 7 to 16-year-old youth with an average body weight (BW) of 45 kg exposed for 10 years. An incidental soil ingestion rate of 100 mg/day is assumed for persons over 6 years old (EPA, 1991). The same value is used for incidental ingestion of sediment, because the activities that result in incidental ingestion of soil and sediment are assumed to be similar. The inhalation rate is assumed to be 13.5 m^3/day (0.56m^3/hour) (EPA, 1997b).

EPA (1997b) identifies a 45-kg youth as being approximately 13 years old. A 13-year-old youth has a total body surface area of about 14,700 cm^2 (EPA, 1992c). Clothing provides partial protection against dermal contact with soil, restricting potential contact to approximately 25 percent of the body, or approximately 3,700 cm^2 (EPA, 1992c). EPA (1992c) recommends a default value of 0.2 mg/cm^2, the lower end of the range of 0.2 to 1.0 mg/cm^2, as an average coefficient for soil-to-skin adherence.

EPA (1989a) permits the development of a fraction term to reflect the proportion of his total daily exposure to soil that a receptor obtains from the contaminated medium. It is assumed that the 4 hours per day that the visitor spends in contact with surface soil, on sites where surface water is present, represents 25 percent of his daily exposure to soil (potential exposure to soil throughout his waking hours, assumed to be 16 hours per day). This percentage is equivalent to a fraction of 0.25, which will be used in the soil ingestion and dermal contact pathways. Similarly, 2 hours or 12.5 percent of the visitor's time (rounded to a fraction of 0.13) is assumed for incidental ingestion and dermal contact with sediment.

During wading, the trespasser or site visitor is assumed to expose his feet, lower legs, hands, and forearms to surface water. These body regions constitute approximately 27 percent of the body surface area of an adult (EPA, 1992c). Assuming that these body regions also constitute 27 percent of the body surface of a 45-kg (13-year-old) youth, the skin surface area exposed to surface water in a wading scenario is estimated at 4,000 cm^2. It is assumed that 25 percent of total body surface area, or about 3,700 cm^2, is exposed to sediment in the wading scenario. The trespasser's or site visitor's contact with surface water is assumed to be intermittent rather than continuous, and it is assumed that uptake of organic chemicals across the dermis does not reach steady state. The default soil-to-skin adherence factor (AF) of 0.2 mg/cm^2 for soil is also used for sediment.

Incidental ingestion of surface water is assumed to occur during wading. An incidental surface water ingestion rate of 0.05 L/hour, the recommended default for swimming (EPA, 2002a), is adopted as the upper bound for wading.

5.3.1.4 Resident

The resident is identified as a possible, although unlikely, future land-use receptor for the landfills and surrounding areas in order to provide a complete risk/hazard evaluation. The resident is assumed to be exposed to total soil by incidental ingestion, dermal contact, inhalation of airborne dust, and inhalation of airborne VOCs. The resident is also exposed to shallow unconfined groundwater (developed as a source of drinking water) by ingestion of drinking water, dermal contact, and inhalation of VOCs. He is also exposed to surface water and sediment by incidental ingestion and dermal contact in the same manner as described for the trespasser.

The resident is assumed to spend some of his time as a child (6 years) and some as an adult (24 years). Residential cancer risk is estimated as the sum of the risks calculated for the child and the adult. This approach captures the greater ingestion and inhalation rate for the child (when normalized to BW) and the longer exposure duration of the adult. Noncancer hazard is estimated for both the adult and child because it is not always clear which receptor provides the more conservative estimate.

The adult resident is assumed to be a 70-kg person with an incidental soil ingestion rate of 100 mg/day, an inhalation rate of 20 m^3/day (0.83m^3/hour), and a drinking water ingestion rate of 2 L/day (EPA, 1991). The adult resident is exposed 350 days per year for 24 years. Approximately 25 percent of his body surface area, or 5,000 cm^2, is available for exposure to soil. EPA (1992c)

recommends a default value of 0.2 mg/cm^2, the lower end of the range of 0.2 to 1.0 mg/cm^2, as an average coefficient for soil-to-skin adherence. The adult resident's entire body surface area, approximately 20,000 cm^2, is available for contact with water during a 0.25-hour-per-day shower (EPA, 1992c). Other occasions for dermal contact with water occur throughout the day, such as hand washing, dish washing, etc. However, the bathing scenario is expected to represent the upper bound; the other occasions are judged to be insignificant by comparison because of the short exposure time and the small part of total body area exposed. Therefore, dermal contact with groundwater used as potable water is quantified only for the bathing scenario.

The child resident is assumed to be a 1- through 6 year-old child with an average BW of 15 kg, a soil ingestion rate of 200 mg/day, an inhalation rate of 12 m^3/day (0.5 m^3/hour), and a drinking water ingestion rate of 1 L/day (EPA, 2002a). The child is exposed 350 days per year (EPA, 1991) for a duration of 6 years (EPA, 2002a). Approximately 25 percent of his body surface area, or 1,700 cm^2, is available for exposure to soil. EPA (1992c) recommends a default value of 0.2 mg/cm^2, the lower end of the range of 0.2 to 1.0 mg/cm^2, as an average coefficient for soil-to-skin adherence. The child resident's entire body, 7,000 cm^2, is available for exposure to groundwater during a 0.25 hour per day bath (EPA, 1992c). Dermal contact with groundwater used as potable water is quantified only for the bathing scenario, as explained above.

The resident may be exposed also to surface water and sediment. This is unlikely to be a regular occurrence for a young child up to 6 years old because caretaker supervision would generally preclude such activity, or reduce it to sporadic and irregular events that do not represent a significant ongoing risk from exposure to contaminants. Older children, however, may spend considerable time playing in surface water. Therefore, the wading scenario assumed for the trespasser or site visitor is adopted for the resident. The risks and hazards from exposure to surface water and sediment estimated for the trespasser will be added to those estimated for residential exposure to soil and potable groundwater. It is recognized that this approach represents a small source of double counting, because the same individual cannot simultaneously be exposed to soil in his own back yard and to surface water and sediment in a pond or ditch. The extent of double counting, however, is small; i.e., of the 8,400 hours the resident is assumed to spend on site (350 days times 24 hours per day), only 104 hours (52 days times 2 hours per day) are spent in contact with surface water and sediment. Therefore, this approach is deemed to be a reasonable approximation for residential exposure.

5.4 Quantification of Exposure-Point Concentrations

The STCs described above are adopted as the EPCs of chemicals in groundwater, soil, surface water, and sediment. EPCs of dust and VOCs in ambient air from soil, of VOCs in household air from use of groundwater as a potable source, of VOCs in ambient air from shallow unconfined groundwater are modeled as described below. Chemical-specific variable values used in the EPC models are compiled in Table M-3.

5.4.1 Exposure-Point Concentrations in Ambient Air

Concentrations of COPCs from Soil Dust. Exposure to airborne dust from soil is a potential exposure pathway for the groundskeeper, construction worker, trespasser or site visitor, and resident. Two phenomena give rise to dust in the air to which a receptor might be exposed:

- Human activity on the site
- Action of the wind.

Airborne dust to which the groundskeeper and construction worker would be exposed is more likely to be raised by the nature of their activities rather than the action of the wind. This is particularly true for the groundskeeper because vegetation markedly reduces wind erosion of soil, and the groundskeeper would spend more time on vegetated than unvegetated soil. Therefore, the most appropriate approach to estimating COPC concentrations in ambient air for the groundskeeper and the construction worker is to use an activity-based dust loading equation (U.S. Department of Energy [DOE], 1989):

$$C_a = (D)(C_s)(CF_1) \qquad \text{Eq. 5-3}$$

where:

C_a	=	chemical concentration in air (milligrams per cubic meter [mg/m^3])
D	=	dust loading factor (grams [g] of soil/m^3 of air)
C_s	=	chemical concentration in soil (mg/kg)
CF_1	=	conversion factor (1E-3 kg/g).

Plausible values for the dust loading factor (D) include 2E-4 g/m^3 for agricultural activity (DOE, 1989), 6E-4 g/m^3 for construction work (DOE, 1983), and 1E-4 g/m^3 for other activity (National Council on Remediation Protection and Measurement [NCRP], 1984). It is assumed that grounds keeping activities are most appropriately reflected by the value of 1E-4 g/m^3 for "other activity," which is used in this evaluation. The value of 6E-4 g/m^3 is used for the construction worker.

The resident, and particularly the trespasser or site visitor, are likely to spend less time operating equipment, so dust is not so likely to be raised by site-related activity as by wind erosion. The extent to which wind raises dust from soil is inversely and linearly related to the extent that the soil is covered with vegetation. EPA (1996a) suggests a conservative default assumption of 50 percent coverage with vegetation. This assumption, although probably overly conservative for the current condition of Landfill Areas 1 and 3, is adopted as sufficiently protective for both the current and the future site-use scenario. The concentration of COPC in air from wind erosion is estimated as follows, based on an "unlimited reservoir" assumption and the assumption that the source area is square:

$$PEF = Q/C \bullet \frac{3600}{0.036 \bullet (1-V) \bullet (U_m / U_t)^3 \bullet F(x)} \qquad \text{Eq. 5-4}$$

where:

PEF	=	particulate emission factor (m^3/kg, calculated)
Q/C	=	inverse of the mean concentration at the center of a square source (50.17 g/square meters (m^2)-second per kg/m^3, or site-specific value from Table 3 in EPA, 1996a, see following paragraph)
0.036	=	respirable fraction (g/m^2-hr)
3,600	=	seconds/hour
V	=	fraction of the surface covered with vegetation (default 0.5 unitless)
U_m	=	mean annual wind speed (default 4.69 m/second)
U_t	=	equivalent threshold value of wind speed at 7 m (default 11.32 m/second)
F(x)	=	function dependent on U_m/U_t (default 0.194).

Q/C is estimated as a site-specific variable using the guidance and data compiled by EPA (1996a). LFNAS is placed in climatic zone VI, and Charleston, South Carolina is selected as most closely approximating meteorological conditions at the Base. Landfill Areas 1 and 3 consist of slightly more than 5 acres each. A Q/C value of 50.17 g/m^2-second per kg/m^3 (based on a 5 acre site) appears to be the most defensible and conservative, and is applicable to both landfills.

The concentration of chemical in the air is calculated as follows:

$$C_a = \frac{C_s}{PEF} \qquad \text{Eq. 5-5}$$

where:

C_a	=	chemical concentration in air (mg/m^3)
C_s	=	chemical concentration in soil (mg/kg)
PEF	=	particulate emission factor (m^3/kg).

Concentrations of COPCs from Soil VOC Volatilization. Exposure to VOCs released from soil is a plausible exposure scenario for any of the receptors hypothetically exposed to surface or total soil. Inhalation exposure to airborne VOCs from soil is evaluated for all receptors.

EPCs of VOCs in ambient air due to volatilization from soil are estimated with a chemical-specific soil volatilization factor calculated from the following equations and defaults provided by EPA (1996a):

$$VF_s = Q/C \bullet CF_2 \bullet \left(\frac{[\,3.14 \bullet D_A \bullet T\,]^{1/2}}{2 \bullet \rho_b \bullet D_A} \right) \qquad \text{Eq. 5-6}$$

and

$$D_A = \frac{(\theta_a^{10/3} \bullet D_i \bullet H' + \theta_w^{10/3} \bullet D_W\,) / n^2}{\rho_b \bullet K_d + \theta_w + \theta_a \bullet H'} \qquad \text{Eq. 5-7}$$

where:

VF_s	=	chemical-from-soil volatilization factor (m^3/kg, chemical-specific, calculated)
Q/C	=	inverse of the mean concentration at the center of the source (50.17 g/m^2-second per kg/m^3, or site-specific value, see above)
CF_2	=	conversion factor (1E-4 m^2/cm^2)
D_A	=	apparent diffusivity (cm^2 per second, calculated)
T	=	exposure interval (seconds: ED · 3.15E7 seconds per year)
ED	=	exposure duration (receptor-specific)
ρ_b	=	dry soil bulk density (1.5 g/cm^3, default, or site-specific)
θ_a	=	air-filled soil porosity (0.28 unitless, default, or site-specific estimated as $n-\theta_w$)
n	=	total soil porosity (0.43 unitless, default, or site-specific estimated as $1-[\rho_b/\rho_s]$)
ρ_s	=	true soil or particle density (2.65 g/cm^3, default, or site-specific)
θ_w	=	water-filled soil porosity (0.15 L_{water}/L_{soil}, default, or site-specific)
D_i	=	diffusivity in air (cm^2/second, chemical specific)
H'	=	dimensionless Henry's law constant (chemical-specific, may be estimated as H · 41)

H	=	Henry's law constant (atmosphere-m^3/mole, chemical-specific)
D_W	=	diffusivity in water (cm^2/second, chemical-specific)
K_d	=	soil-water partition coefficient (cubic centimeters per gram [cm^3/g], chemical-specific, may be estimated as $K_{oc} \cdot f_{oc}$)
K_{oc}	=	soil organic carbon-water partition coefficient (cm^3/g, chemical-specific)
f_{oc}	=	organic carbon content of soil (g/g, site-specific).

The concentration of chemical in the air is calculated as follows:

$$C_a = \frac{C_s}{VF_s} \qquad \text{Eq. 5-8}$$

where:

C_a	=	chemical concentration in air (mg/m^3)
C_s	=	chemical concentration in soil (mg/kg)
VF_s	=	chemical-from-soil volatilization factor (m^3/kg).

Concentrations of COPCs from shallow groundwater near the surface. It is assumed that construction worker exposure to shallow groundwater may occur during excavation and work in a utilities trench that is 2 feet (51 cm) wide, 2.5 feet (76 cm) deep, and of undetermined length. Inhalation exposure to airborne VOCs from the shallow unconfined groundwater near the ground surface will be quantified.

Two separate models are available to estimate airborne concentrations of VOCs. The ASTM (1995) model would be used when depth to groundwater equals or exceeds 81 cm. Visual appraisal of the data in Appendix M, however, indicates that the depth to groundwater may well fall short of 81 cm, particularly during wet seasons, and it is likely that groundwater could pool in the hypothetical utilities trench. Under these conditions, a factor is used to model diffusive mass transfer from water to air (McKone, 1993), and box-model dilution is used to estimate the concentration in ambient air:

$$C_a = \frac{(T_{sw-air})(A)(C_g)(CF_3)}{(U_m)(\delta_{air})} \qquad \text{Eq. 5-9}$$

where:

C_a	=	contaminant concentration in air (mg/m^3, calculated)
$T_{sw\text{-}air}$	=	surface water-to-air diffusive mass transfer factor ($seconds^{-1}$)
A	=	surface area of pooled groundwater (9,290 cm^2, see below)
C_g	=	concentration of VOC in groundwater (mg/L)
CF_3	=	conversion factor (1E+3 L/m^3)

U_m	=	average wind speed above ground surface (site-specific, or default 469 cm/second [EPA, 1996a])
δ_{air}	=	ambient air mixing zone height (200 cm [ASTM, 1995]).

It is assumed that the groundwater pool covers a 5 feet (152 cm) by 2 feet (61 cm) section, which is equivalent to 9,290 cm^2.

The surface water-to-air diffusive mass transfer factor is estimated as (McKone, 1993):

$$T_{sw-air} = \frac{1}{\left(\frac{H/RT}{U_{sw}} + \frac{1}{U_{air}}\right)\delta_{air}} \qquad \text{Eq. 5-10}$$

and

$$U_{air} = \frac{D_i}{\delta_{sw}} \qquad \text{Eq. 5-11}$$

where:

$T_{sw\text{-}air}$	=	surface water-to-air diffusive mass transfer factor ($seconds^{-1}$, calculated)
H	=	Henry's Law constant (atmosphere-m^3/mole, chemical-specific)
R	=	gas law constant (8.2E-5 atmosphere-m^3/mole-°K)
T	=	ambient temperature (293.7°K, see below)
U_{sw}	=	mass transfer coefficient on surface water side of air-surface water boundary (2.7E-4 cm/second, McKone, 1993)
U_{air}	=	mass transfer coefficient on air side of air-surface water boundary (cm/second, calculated)
δ_{air}	=	ambient air mixing zone height (200 cm [ASTM, 1995])
D_i	=	diffusivity in air (chemical-specific, cm^2/second)
δ_{sw}	=	boundary layer thickness in air above surface water (1 cm, McKone, 1993).

The value for T of 293.7° K is estimated from the mean temperature for Jacksonville, Florida of 68.8 °F (Southeast Regional Climate Center [SERCC], 2003).

Concentration of COPCs from Potable Water VOC Volatilization. Two approaches are generally used to evaluate residential inhalation exposure to VOC emissions from potable water from household use. One is to consider a showering scenario as the upper bound, because warm flowing water divided by the showerhead results in maximum volatilization, and the source of airborne VOCs and the receptor are confined within a relatively small space with limited air

exchange. Only VOC inhalation from the showering scenario is quantified. A major limitation of this approach is the assumption that adults shower, but young children are more likely to take baths; therefore, inhalation exposure is not quantified for the child.

The other approach is to use a whole-house model that estimates airborne concentrations of VOCs from all household uses of potable water. This approach, which permits quantification of this pathway for the child resident as well as the adult, is adopted for this evaluation.

California Environmental Protection Agency (Cal-EPA) (1992) recommends a simple model to relate the average daily concentration of VOC in indoor air with the concentration in potable water:

$$C_{house} = \frac{W_{house} \bullet \phi_x \bullet C_{pw}}{VR_{house}} \qquad \text{Eq. 5-12}$$

where:

C_{house}	=	concentration of VOC in indoor air, daily average for entire house (mg/m^3).
W_{house}	=	average daily household indoor water use (447 L/day, see below).
ϕ_x	=	VOC mass transfer coefficient, all household uses, (0.54, unitless [Cal-EPA, 1992])
C_{pw}	=	concentration of VOC in potable water (mg/L)
VR_{house}	=	average daily ventilation rate (3,984 m^3/day, see below).

EPA (1997b) reviewed several studies of the volumes of houses and air exchange rates, and recommends 369 m^3 as a central estimate of the volume of a house, and 0.45 air changes per hour as a typical air exchange rate. A ventilation rate for a typical house of 166 m^3/hour (3,984 m^3/day) is estimated from these values. EPA (1997b) also related residential volumes to household size. The average volume of a 1-person household is 269 m^3; the average volume of a 2-person household is 386 m^3. The average volume of all households, 369 m^3 as described above, is similar to the volume for a 2-person household. Therefore, it is assumed that the average household contains two persons. EPA (1997b) estimated a mean per capita indoor water use rate of 59 gallons/day. Assuming a 2-person household, this estimate is equivalent to 118 gallons/day, or 447 L/day.

5.4.2 Future Groundwater Conditions Exposure-Point Concentrations

Groundwater at the LFNAS Landfill Areas 1 and 3 is shallow, suggesting that there is little impediment to leaching to groundwater. Also, the wide range of chemicals detected in

groundwater suggests that maximum leaching may have already occurred. Therefore, it is judged that the current groundwater evaluation captures the extent of potential groundwater contamination and that modeling EPCs for future groundwater conditions is unnecessary.

5.5 Quantification of Chemical Intake

This section describes the models used to quantify dose or uptake rates of the COPCs by the receptors and exposure pathways previously identified. Most risk assessments are based on a reasonable maximum exposure (RME) assumption. The intent of the RME assumption is to estimate the highest exposure level that could reasonably be expected to occur, but not necessarily the worst possible case (EPA, 1989a, 1991). It is interpreted as reflecting the 90th to 95th percentile of exposure. In keeping with EPA guidance (1991), variables chosen for a baseline RME scenario for intake rate, exposure frequency (EF),and exposure duration (ED) are generally upper bounds. Other variables, i.e., BW and exposed skin surface area (SA), are generally central or average values. In the case of contact rates consisting of multiple components, e.g., dermal contact with soil or water, which consists of a dermal absorption factor (ABS) and soil-to-skin AF for soil, and permeability coefficient (K_p) and exposure time (ET) for water, only one variable, ABS or K_p needs to be an upper bound. The conservatism built into the individual variables ensures that the entire estimate for contact rate is more than sufficiently conservative.

The averaging time (AT) for noncancer evaluation is computed as the product of ED (years) times 365 days/year, to estimate an average daily dose rate over the entire exposure period (EPA, 1989a). For cancer evaluation, AT is computed as the product of 70 years, the assumed human lifetime, times 365 days/year, to estimate an average daily dose prorated over a lifetime, regardless of the frequency or duration of exposure. This methodology assumes that the risk from short-term exposure to a high dose of a given carcinogen is equivalent to long-term exposure to a correspondingly lower dose, provided that the total lifetime doses are equivalent. This approach is consistent with current EPA (1986b) policy of carcinogen evaluation, although it introduces considerable uncertainty into the cancer risk assessment.

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

soil and sediment, FIs are introduced that apportion his exposure between the two media. The default value of FI is 1.

Models were taken or modified from EPA (1989a) unless otherwise indicated. The exposure variable values used in the following models are compiled in Table M-2, chemical-specific variable values are compiled in Table M-3.

5.5.1 Inhalation of COPCs in Air

Inhalation of Airborne COPCs from Soil or Water. The groundskeeper, construction worker, trespasser, and resident are subject to inhalation of COPCs from airborne dust and inhalation of VOC emissions from soil. The construction worker is also subject to inhalation of VOCs from shallow groundwater while working in a trench. The resident is also subject to inhalation of VOCs from groundwater in household air. The inhaled dose of a COPC in air is estimated as follows:

$$I_a = \frac{(C_a)(IN)(FI)(ET)(EF)(ED)}{(BW)(AT)} \qquad \text{Eq. 5-13}$$

where:

I_a	=	inhaled dose of COPC (mg/kg-day)
C_a	=	concentration of COPC in air (mg/m^3)
IN	=	(IN_{so} for COPCs from soil; IN_{pw} for COPCs from potable water) inhalation rate (m^3/hour)
FI	=	fraction of exposure attributed to site medium (unitless)
ET	=	(ET_{so} for COPCs from soil; ET_{pw} for COPCs from potable water) exposure time (hours/day)
EF	=	exposure frequency (days/year)
ED	=	exposure duration (years)
BW	=	body weight (kg)
AT	=	averaging time (days).

5.5.2 Incidental Ingestion of COPCs in Soil or Sediment

Incidental soil ingestion is a plausible exposure pathway for the groundskeeper, construction worker, trespasser, and resident. The trespasser (site visitor) and resident are also subject to incidental ingestion of sediment. The ingested dose of COPC in soil or sediment is estimated from the equation:

$$I_s = \frac{(C_s)(IR)(FI)(EF)(ED)(CF_4)}{(BW)(AT)} \qquad \text{Eq. 5-14}$$

where:

I_s	=	incidental ingested dose of COPC in soil (mg/kg-day)
C_s	=	concentration of COPC in soil (mg/kg)
IR	=	(IR_{so} for COPCs in soil; IR_{sd} for COPCs in sediment) ingestion rate (mg/day)
FI	=	(FI_{so} for COPCs in soil; FI_{sd} for COPCs in sediment) fraction of exposure attributed to site soil (unitless)
EF	=	exposure frequency (days/year)
ED	=	exposure duration (years)
CF_4	=	conversion factor (1E-6 kg/mg)
BW	=	body weight (kg)
AT	=	averaging time (days).

5.5.3 Ingestion of COPCs in Water

Ingestion of COPCs in Groundwater Developed as a Potable Source. Ingestion of drinking water is a potential exposure route for the groundskeeper, construction worker, and resident. The ingested dose of COPC in drinking water is estimated as follows:

$$I_{pw} = \frac{(C_{pw})(IR_{pw})(FI_{pw})(EF)(ED)}{(BW)(AT)} \qquad \text{Eq. 5-15}$$

where:

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

Ingestion of COPCs in Surface Water. Equation 15 is modified slightly as follows to quantify incidental ingestion of surface water by the trespasser or visitor as follows:

$$I_{sw} = \frac{(C_{sw})(IR_{sw})(ET_{sw})(EF)(ED)}{(BW)(AT)} \qquad \text{Eq. 5-16}$$

where:

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

5.5.4 Dermal Uptake of COPCs from Soil, Sediment, or Water

Unlike the methodologies for estimating inhaled or ingested dose of COPCs, which quantify the dose presented to the barrier membrane (the pulmonary or gastrointestinal mucosa, respectively), dermal dose is estimated as the dose that crosses the skin and is systemically absorbed. For this reason, dermal toxicity values are also based on absorbed dose. The absorbed dose of COPCs for the groundskeeper, construction worker, trespasser, and adult and child resident is estimated as follows (EPA, 1992c):

$$DAD = \frac{(DA)(SA)(FI)(EF)(ED)}{(BW)(AT)} \qquad \text{Eq. 5-17}$$

where:

DAD	=	average dermally absorbed dose of COPC (mg/kg-day)
DA	=	dose absorbed per unit body surface area per day (mg/cm^2)
SA	=	surface area of the skin available for contact with site medium (cm^2/day)
FI	=	fraction of exposure attributed to site medium (unitless)
EF	=	exposure frequency (days/year)
ED	=	exposure duration (years)
BW	=	body weight (kg)
AT	=	averaging time (days).

It is assumed that one exposure event occurs each exposure day to maintain the dimensional integrity of the equation.

DA Calculation for Soil or Sediment. The dose absorbed (DA) is calculated differently for dermal uptake from soil or sediment than from water. Dermal uptake of COPCs from soil or sediment assumes that absorption is a function of the fraction of a dermally applied dose that is absorbed. It is calculated from the following equation (EPA, 1992c):

$$DA = (C\ )(CF_4)(AF)(ABS) \qquad \text{Eq. 5-18}$$

where:

DA	=	dose absorbed per unit body surface area per day (mg/cm^2)
C	=	(C_{so} for COPCs in soil; C_{sd} for COPCs in sediment) concentration of COPC (mg/kg)
CF_4	=	conversion factor (1E-6 kg/mg)
AF	=	soil- or sediment-to-skin adherence factor (mg/cm^2)
ABS	=	absorption fraction (unitless, chemical-specific value).

DA Calculation for Water. Separate calculation methods are applied to estimate DA for inorganic and organic chemicals in water. For inorganic chemicals, DA is calculated from the following equation:

$$DA = (C\ )(K_p)(ET)(CF_5) \qquad \text{Eq. 5-19}$$

where:

DA	=	dose absorbed per unit body surface area per day (mg/cm^2)
C	=	(C_{pw} for COPCs in potable water; C_{sw} for COPCs in surface water) concentration of COPC in water (mg/L)
K_p	=	permeability coefficient (cm/hour)
ET	=	(ET_{pw} for COPCs in potable water; ET_{sw} for COPCs in surface water) dermal exposure time (hours)
CF_5	=	conversion factor (0.001 liters per cubic centimeter [L/cm^3])

Quantification of dermal uptake of COPC from water depends on a K_p, which describes the rate of movement of a chemical from water across the dermal barrier to the systemic circulation (EPA, 1992c). Dermal contact with groundwater applies to the groundskeeper, construction worker, and child and adult resident. The equation for dermal uptake of chemicals from water is the same as the equation for dermal uptake of chemicals from soil or sediment (Equation 5-17).

DA for Organic Chemicals in Water. K_p for organic chemicals varies by several orders of magnitude and is highly dependent on lipophilicity, expressed as a function of the octanol/water partition coefficient (K_{ow}) (EPA, 1992c). Because the stratum corneum (the outer skin layer) is rich in lipid content, it may act as a sink, initially reducing the transport of chemical to the systemic circulation. With continued exposure and the attainment of steady-state conditions, the rate of dermal uptake increases. Therefore, different equations are used to estimate DA, depending on whether the exposure time is less than or greater than the estimated time to reach

steady state. When steady state has not been reached, which is assumed to be the case for the receptors evaluated at LFNAS, the following equation is applied (EPA, 1992c):

$$DA = 2(K_p)(C)(CF_5)\sqrt{\left(\frac{6\tau(ET)}{\pi}\right)} \qquad \text{Eq. 5-20}$$

where:

DA	=	dose absorbed per unit body surface area per day (mg/cm^2)
K_p	=	permeability coefficient (cm/hour)
C	=	(C_{pw} for COPCs in potable water; C_{sw} for COPCs in surface water) chemical concentration in water (mg/L)
CF_5	=	conversion factor (0.001 L/cm^3)
τ	=	time for concentration of contaminant in stratum corneum to reach steady state (hours)
ET	=	(ETpw for COPCs in potable water; ET_{sw} for COPCs in surface water) dermal exposure time (hours/day).

5.6 Toxicity Assessment

Toxicity is defined as the ability of a chemical to induce adverse effects in biological systems. The purpose of the toxicity assessment is two-fold:

- Identify the cancer and noncancer effects that may arise from exposure of humans to the COPCs (hazard assessment).
- Provide an estimate of the quantitative relationship between the magnitude and duration of exposure and the probability or severity of adverse effects (dose-response assessment).
- The latter is accomplished by the derivation of cancer and noncancer toxicity values, as described in the following subsection.

5.6.1 Evaluation of Cancer Risk

A few chemicals are known to be human carcinogens, and many more are suspected to be human carcinogens. The evaluation of the potential carcinogenicity of a chemical includes both a qualitative and a quantitative aspect (EPA, 1986b). The qualitative aspect is a weight-of-evidence evaluation of the likelihood that a chemical might induce cancer in humans. EPA (1986b) recognizes six weight-of-evidence group classifications for carcinogenicity:

- ***Group A - Human Carcinogen.*** Human data are sufficient to identify the chemical as a human carcinogen.

- ***Group B1 - Probable Human Carcinogen.*** Human data indicate that a causal association is credible, but alternative explanations cannot be dismissed.

- ***Group B2 - Probable Human Carcinogen.*** Human data are insufficient to support a causal association, but testing data in animals support a causal association.

- ***Group C - Possible Human Carcinogen.*** Human data are inadequate or lacking, but animal data suggest a causal association, although the studies have deficiencies that limit interpretation.

- ***Group D - Not Classifiable as to Human Carcinogenicity.*** Human and animal data are lacking or inadequate.

- ***Group E - Evidence of Noncarcinogenicity to Humans.*** Human data are negative or lacking, and adequate animal data indicate no association with cancer.

More recently, EPA (1999) proposed a less formal and more narrative description of cancer weight of evidence as follows:

Carcinogenicity

- Carcinogenic to humans.

- Likely to be carcinogenic to humans.

- Suggestive evidence of carcinogenicity, but not sufficient to assess human carcinogenic potential.

- Data are inadequate for an assessment of human carcinogenic potential.

- Not likely to be carcinogenic to humans.

The toxicity value for carcinogenicity, called a cancer slope factor (SF), is an estimate of potency. Potency estimates are developed only for chemicals in Groups A, B1, B2, and C, and only if the data are sufficient. The potency estimates are statistically derived from the dose-response curve from the best human or animal study or studies of the chemical. Although human data are often considered to be more reliable than animal data because there is no need to extrapolate the results obtained in one species to another, most human studies have one or more of the following limitations:

- The duration of exposure is usually considerably less than lifetime.

- The concentration or dose of chemical to which the humans were exposed can be only crudely approximated, usually from historical data.

- Concurrent exposure to other chemicals frequently confounds interpretation.

- Data regarding other factors (tobacco, alcohol, illicit or medicinal drug use, nutritional factors and dietary habits, heredity) are usually insufficient to eliminate confounding or quantify its effect on the results.

- Most epidemiologic studies are occupational investigations of workers, which may not accurately reflect the range of sensitivities of the general population.

- Most epidemiologic studies lack the statistical power (i.e., sample size) to detect a low, but chemical-related increased incidence of tumors.

Most potency estimates are derived from animal data, which present different limitations:

- It is necessary to extrapolate from results in animals to predict results in humans, usually done by estimating an equivalent human dose rate from the animal dose rate.

- The range of sensitivities arising from genotypic and phenotypic diversity in the human population is not reflected in the animal models ordinarily used in cancer studies.

- Usually very high doses of chemical are used, which may alter normal biology, creating a physiologically artificial state and introducing substantial uncertainty regarding the extrapolation to the low-dose range expected with environmental exposure.

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

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

$$(p_{(d)} - p_{(0)}) / (1 - p_{(0)}) \qquad \text{Eq. 5-21}$$

where:

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

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

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

The EPA (2003) Integrated Risk Information System (IRIS) expresses inhalation cancer potency as a unit risk-based on concentration, or risk per microgram of chemical per cubic meter of ambient air ($\mu g/m^3$). Because cancer risk characterization requires potency expressed as risk per mg/kg-day, the unit risk must be converted to the mathematical equivalent of an inhalation cancer SF, or risk per unit dose. Since the inhalation unit risk is based on continuous lifetime exposure of an adult human (assumed to inhale 20 m^3 of air per day and to weigh 70 kg) the mathematical conversion consists of multiplying the unit risk (per $\mu g/m^3$) by 70 kg and by 1,000 $\mu g/mg$, and dividing the result by 20 m^3/day.

5.6.2 Evaluation of Noncancer Effects

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

- Qualitative identification of the adverse effect(s) associated with the chemical; these may differ depending on the duration (acute or chronic) or route (oral or inhalation) of exposure.
- Identification of the critical effect for each duration of exposure (i.e., the first adverse effect that occurs as dose is increased above zero).
- Estimation of the threshold dose for the critical effect for each duration of exposure.
- Development of an uncertainty factor, i.e., quantification of the uncertainty associated with interspecies extrapolation, intraspecies variation in sensitivity, severity of the critical effect and slope of the dose-response curve, and deficiencies in the database, in regard to developing a RfD for human exposure.
- Identification of the target organ for the critical effect for each route of exposure.

These information points are used to derive an exposure route- and duration-specific toxicity value called an RfD, expressed as mg/kg-day, which is considered to be the dose for humans, with uncertainty of an order of magnitude or greater, at which adverse effects are not expected to occur. Mathematically, it is estimated as the ratio of the threshold dose to the uncertainty factor. For purposes of risk assessment, chronic exposure is defined as equal to or greater than 7 years, i.e., at least 10 percent of expected lifespan; subchronic exposure is defined as 2 weeks to 7 years.

IRIS (EPA, 2003) and the Health Effects Assessment Summary Tables (HEAST) (EPA, 1997c) express the inhalation noncancer reference value as a reference concentration (RfC) in units of mg/m^3. Because noncancer risk characterization requires a reference value expressed as mg/kg-day, the RfC must be converted to an inhalation RfD. Since the inhalation RfC is based on continuous exposure of an adult human (assumed to inhale 20 m^3 of air/day and to weigh 70 kilograms), the mathematical conversion consists of multiplying the RfC (mg/m^3) by 20 m^3/day and dividing the result by 70 kilograms.

Iron. Iron was identified as a COPC in groundwater at Landfill Area 1, and in groundwater and sediment at Landfill Area 3. The following is adapted from the toxicity profile for iron. Iron is a nutritionally required trace element that forms an integral part of hemoglobin, myoglobin and several enzymes. Chronic iron toxicity is seldom observed in humans with normal homeostatic control of iron absorption. The appropriateness of the oral RfD for iron is controversial, especially for use with exposure to iron in soil or sediment. The oral RfD documentation reviewed in the toxicity profile reflects estimates of iron intakes based on nutritional surveys that identify neither a no-observed-adverse-effect level nor a lowest-observed-adverse-effect level for iron ingestion. In other words, it is entirely possible that a much higher level of dietary iron intake would be innocuous, although this possibility is not evaluated in the oral RfD documentation.

In addition, it is likely that the forms of iron present in the plant and animal products that comprise the diets evaluated in the nutritional survey are more bioavailable than those present in soil, sediment, or groundwater.

There are only two known instances of toxicity to iron following oral exposure. One involves acute ingestion by children of large quantities of iron-containing nutritional supplements especially formulated to have a pleasing taste and to enhance bioavailability for their medicinal

effects. The other is the occurrence of hemosiderosis in Bantu consuming Kaffir beer. It has been established, however, that the iron in the beer is present in a soluble form that is fully as bioavailable as the iron in ferric chloride. Presumably, high bioavailability is important in development of toxicity to ingested iron. Therefore, it is likely that the oral RfD for iron exaggerates noncancer human hazards beyond the conservative assumptions typically associated with EPA toxicity values. For these reasons, consistent with EPA (2002a) guidance, HI values are not estimated for iron selected as a COPC in environmental media at the former landfills.

Target Organ Toxicity. As a matter of science policy, EPA assumes dose- and effect-additivity for noncancer effects (EPA, 1989a). This assumption provides the justification for adding the hazard quotients (HQ) or HIs in the risk characterization for noncancer effects resulting from exposure to multiple chemicals, pathways, or media. EPA (1989a), however, acknowledges that adding all HQ or HI values may overestimate risk, because the assumption of additivity is probably appropriate only for those chemicals that exert their toxicity by the same mechanism.

Data regarding the mechanism of toxicity sufficient for predicting additivity with a high level of confidence are available for very few chemical combinations. In the absence of such data, EPA (1989a) assumes that chemicals that act on the same target organ may do so by the same mechanism of toxicity, i.e., the target organ serves as a surrogate for mechanism of toxicity for determining which HI values to sum. When total HI summed across all media for a receptor exceeds the threshold level of 1 due to the contributions of several chemicals, it may be appropriate to segregate the chemicals by route of exposure and mechanism of toxicity (i.e., target organ) and estimate separate HI values for each.

As a practical matter, since human environmental exposures are likely to involve near- or sub-threshold doses, the target organs chosen for a given chemical are the ones associated with effects at or near the threshold. The target organ is also selected on the basis of duration of exposure (i.e., the target organ for chronic or subchronic exposure to low or moderate doses is selected rather than the target organ for acute exposure to high doses) and route of exposure. Because dermal RfD values are derived from oral RfD values, the oral target organ is adopted as the dermal target organ. For some chemicals, no target organ is identified. This may arise when no adverse effects are observed or when adverse effects such as reduced longevity or growth rate are not accompanied by recognized organ- or system-specific functional or morphologic alteration.

Dermal Toxicity Values. Dermal RfD values and SFs are derived from the corresponding oral values, provided there is no evidence to suggest that dermal exposure induces exposure route-specific effects that are not appropriately modeled by oral exposure data. In the derivation of a dermal RfD, the oral RfD is multiplied by the gastrointestinal absorption factor (GAF), expressed as a decimal fraction. The resulting dermal RfD, therefore, is based on absorbed dose. The RfD based on absorbed dose is the appropriate value with which to compare a dermal dose, because dermal doses are expressed as absorbed rather than exposure doses. The dermal SF is derived by dividing the oral SF by the GAF. The oral SF is divided, rather than multiplied, by the GAF because SFs are expressed as a reciprocal dose.

5.6.3 Sources of Toxicity Information Used in the Risk Assessment

Toxicity values are chosen using the following hierarchy:

- The EPA's on-line IRIS database (EPA, 2003) containing toxicity values that have undergone the most rigorous agency review.
- The latest version of the annual HEAST, including all supplements (EPA, 1997c).
- Other EPA documents, memoranda, or National Center for Environmental Assessment (NCEA) derivations for the Superfund Technical Support Center. There may be chemicals for which the NCEA assessments are more recent or for other reasons (e.g., newer empirical data were evaluated) will be considered to supercede assessments in EPA (1997c, 2003).

All toxicity values, regardless of their source, are evaluated for appropriateness for use in risk assessment. When toxicity values are not located, the primary literature may be surveyed in order to determine if sufficient data exist that would permit the derivation of a toxicity value. The use of surrogate chemicals is also considered, if the chemical structure, adverse effects, and toxic potency of the surrogate and chemical of interest are judged to be sufficiently similar.

Gastrointestinal Absorption Factors. GAFs, used to derive dermal RfD values and SFs from the corresponding oral toxicity values, are obtained from the following sources:

- EPA (2001) guidance
- Empirical data
- NCEA position papers

- Federal agency reviews of the empirical data, such as Agency for Toxic Substances and Disease Registry Toxicological Profiles and various EPA criteria documents

- Other published reviews of the empirical data.

GAFs obtained from reviews are compared to empirical (especially more recent) data, when possible, and are evaluated for suitability for use for deriving dermal toxicity values from oral toxicity values. The suitability of the GAF increases when the following similarities are present in the oral pharmacokinetic study from which the GAF is derived and in the key toxicity study from which the oral toxicity value is derived:

- The same strain, sex, age, and species of test animal was used.

- The same chemical form (e.g., the same salt or complex of an inorganic element or organic compound) was used.

- The same mode of administration (e.g., diet, drinking water, or gavage vehicle) was used.

- Similar dose rates were used.

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

5.6.4 Toxicological Profiles

Toxicological profiles have been developed for all chemicals selected as COPC (Appendix M). The profile includes data such as the general use of the chemical, synonyms, CAS number, chemical-specific physical characteristics, and current and available toxicological and toxicokinetic information (RfDs, SFs, and GAFs). Table M-4 summarizes the chemical-specific toxicological data for all COPCs.

5.7 Risk Characterization

Risk characterization is the combination of the results of the exposure assessment and toxicity assessment to yield a quantitative expression of risk for the exposed receptors. This quantitative expression is the probability of developing cancer, or a nonprobabilistic comparison of estimated dose with an RfD for noncancer effects. Quantitative estimates are developed for individual chemicals, exposure pathways, and exposure media for each receptor. Although some chemicals induce both cancer and noncancer effects, the risks for each type of effect are calculated separately. The risk characterization is used to guide risk management decisions.

Generally, the risk characterization follows the methodology prescribed by EPA (1989a). The EPA methods are designed to be health-protective, and tend to overestimate rather than underestimate risk.

5.7.1 Cancer Risk

The risk of developing cancer from exposure to potential chemical carcinogens is estimated as the probability of an individual developing cancer over a lifetime. In the low-dose range, which would be expected for most environmental exposures, cancer risk is estimated from the following linear equation (EPA, 1989a):

$$ILCR = (CDI)(SF) \qquad \text{Eq. 5-22}$$

where:

ILCR	=	incremental lifetime cancer risk, a unitless expression of the probability of developing cancer, adjusted for background incidence
CDI	=	chronic daily intake, averaged over the expected human lifetime (70 years) (mg/kg-day)
SF	=	cancer slope factor (per mg/kg-day).

The chronic daily intake (CDI) term in Equation 5-22 is equivalent to the "I" or "DAD" terms (intake or dose rates) in Equations 5-13 through 5-17 when these equations are evaluated for cancer intakes.

The use of the above equation assumes that chemical carcinogenesis does not exhibit a threshold, and that the dose-response relationship is linear in the low dose range; because this equation could generate theoretical cancer risks greater than 1 for high dose levels, it is considered to be inaccurate at cancer risks greater than 1E-2. In these cases, cancer risk is estimated by the one-hit model (EPA, 1989a):

$$ILCR = 1 - e^{[(-CDI)(SF)]} \qquad \text{Eq. 5-23}$$

where:

ILCR	=	incremental lifetime cancer risk, a unitless expression of the probability of developing cancer, adjusted for background incidence
$e^{(-CDI)(SF)}$	=	the exponential of the negative of the risk calculated using previous equation.

As a matter of policy, EPA (1986b) considers the carcinogenic potency of simultaneous exposure to low doses of carcinogenic chemicals to be additive, regardless of the chemical's mechanisms of toxicity or sites (organs of the body) of action. Cancer risk arising from simultaneous exposure by a given pathway to multiple chemicals is estimated from the equation (EPA, 1989a):

$$Risk_p = ILCR_{(chem\ 1)} + ILCR_{(chem\ 2)} + \cdots ILCR_{(chem\ i)} \qquad \text{Eq. 5-24}$$

where:

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

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

5.7.2 Noncancer Effects of Chemicals

The risk associated with the noncancer effects of a chemical is more appropriately termed hazard (to distinguish it from cancer risk, which is expressed as a numerical probability), and is evaluated by comparing an exposure level or intake with a RfD. The HQ, defined as the ratio of intake to RfD, is defined as follows (EPA, 1989a):

$$HQ = I \,/\, RfD \qquad \text{Eq. 5-25}$$

where:

HQ	=	hazard quotient (unitless)
I	=	intake of chemical (mg/kg-day)
RfD	=	reference dose (mg/kg-day).

The I term in Equation 5-25 is equivalent to the "I" or "DAD" terms (intake or dose rates) in Equations 5-13 through 5-17 when these equations are evaluated for noncancer intakes.

This approach is different from the probabilistic approach used to evaluate carcinogenic effects. An HQ of 0.01 does not imply a 1 in 100 chance of an adverse effect, but indicates that the estimated intake is 100 times lower than the RfD. An HQ of unity indicates that the estimated intake equals the RfD. If the HQ is greater than unity, there may be concern for potential adverse health effects.

In the case of simultaneous exposure of a receptor to several chemicals, an HI for a given exposure pathway is calculated as the sum of the HQs by:

$$HI = HQ_1 + HQ_2 + \ldots HQ_i \qquad \text{Eq. 5-26}$$

where:

HI	=	hazard index (unitless)
HQ	=	hazard quotient for the ith toxicant

An HI for multiple exposure by multiple pathways and across multiple media can be summed in the same manner.

If the HI for a given exposure pathway exceeds 1.0, individual HI values may be calculated for each target organ.

5.8 Remedial Goal Option Development

Region 4 requires development of RGOs as part of the BHHRA (EPA, 2002b). RGOs are site-specific risk-based concentrations that reflect the exposure and toxicity assumptions applied in the BHHRA. Consequently, the risk-based RGOs are source medium-, receptor-, and chemical-specific.

The first step in RGO development is selection of COCs. Either of two conditions results in the designation of a COPC as a COC:

- The concentration of the COPC exceeds its medium-specific ARAR.
- The COPC contributes significantly to a total receptor incremental lifetime cancer risk (ILCR) greater than 1E-6 or a total receptor HI greater than 1.

Significant contribution to cancer risk for a specific chemical is defined as contributing an ILCR across all exposure pathways for a given source medium exceeding 1E-6; significant contribution to hazard for a specific chemical is defined as contributing an HI across all exposure pathways for a given source medium exceeding 0.1. The COC, therefore, may be selected because of its cancer risk (cancer COC) or noncancer hazard (noncancer COC) or both.

RGOs for cancer COCs are estimated for a given medium from the following equation:

$$RGO_{coc} = \frac{ST_{coc}\ TR}{ILCR_{COC}} \qquad \text{Eq. 5-27}$$

where:

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

RGOs for noncancer COC are estimated as follows:

$$RGO_{coc} = \frac{ST_{coc}\ THI}{HI_{COC}} \qquad \text{Eq. 5-28}$$

where:

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

Concentration units are not provided in the RGO equations; the RGO units will be the same as the concentration units of the source-term concentration for each medium.

5.9 Human Health Risk Assessment Results

This section summarizes the results of Phase I (i.e., COPC selection) and Phase II (i.e., BHHRA) of the risk assessment tasks. Generally, cancer risk estimates below or equal to 1E-6 are considered negligible (EPA, 1990). A cancer risk of 1E-6 is considered a point of departure, above which concern rises. An HI below or equal to the threshold level of 1.0 is considered acceptable; above 1.0 indicates concern about the occurrence of adverse noncancer effects (EPA, 1989a).

EPA (1989a; 2002a) suggests rounding risk estimates to one significant figure to reflect the uncertainty inherent in their estimation. Numerical estimates in the tables will be presented in scientific notation with one integer and two digits to the right of the decimal. This presentation facilitates document review and replication of the calculations. Rounding of risk estimates will be done only in the text, and only when it is helpful for interpretation. For example, an HI of

1.49E+0 would be rounded to 1 and interpreted as equal to, but not exceeding, the threshold level of 1.

5.9.1 Landfill Area 1

5.9.1.1 Assessment of Contamination

Surface soil, total soil, groundwater, surface water, and sediment samples taken from Landfill Area 1 are identified in Tables M-5, M-6, M-7, M-8, and M-9, respectively. All media were analyzed for metals, PCBs, organochlorine pesticides, VOCs, SVOCs, and TRPH, although not each sample was analyzed for all chemical classes.

A total of 20 surface soil samples were taken – 15 from within the boundaries of the landfill and 5 from outside the boundaries of the landfill (Table M-5). COPCs identified in surface soil (Tables M-10 and M-11) include a low concentration of a single metal (antimony), PCBs, a low concentration of a single organochlorine pesticide (dieldrin), several PAHs, and TRPH. There appears to be little difference between surface soil inside and outside the landfill boundary. A total of seven samples of subsurface soil were taken: 3 from within the boundaries of the landfill and 4 from outside the boundaries of the landfill (Table M-5). COPCs in total soil (Table M-12) were similar to those in surface soil, but some PCB and PAH concentrations were slightly higher in subsurface soil than in surface soil. This is counterintuitive because these chemicals are relatively immobile in soil and tend not to migrate to deeper soil horizons. VOCs were present in soil in relatively few samples at low concentrations; none were selected as COPCs.

A total of 22 groundwater samples were taken and analyzed as described above (Table M-7). COPCs identified in groundwater include two metals (iron, manganese), one SVOC (bis[2-ethylhexyl]phthalate), two VOCs (chlorobenzene, vinyl chloride), and TRPH (Table M-13). A total of four VOCs were identified in groundwater. Three were identified in only one of 22 samples. Vinyl chloride was identified in three of 22 samples. Of the four VOCs identified in groundwater, only one (cymene) was identified in soil. TRPH was a consistent finding in both soil and groundwater.

Four surface water and sediment samples were taken and analyzed as described above (Tables M-8 and M-9). Chemical detections in surface water were limited to metals and TRPH (Table M-14). TRPH was selected as the only COPC in surface water. Chemicals detected in sediment include metals, PCBs, an organochlorine pesticide, VOCs, and TRPH (Table M-15). These

chemicals were present at levels sufficiently low that none were selected as COPCs. Therefore, sediment at Landfill Area 1 was not evaluated in the BHHRA.

5.9.1.2 Risk Assessment and RGO Development

Four receptor scenarios – groundskeeper, construction worker, trespasser or site visitor, and resident – were evaluated for exposure to media at Landfill Area 1. The media to which each receptor was assumed to be exposed is presented in Section 5.1.5.1 and Table M-1. Risk estimates for each COPC, and risk estimates summed across all COPCs for a given medium, are compiled for surface soil (Tables M-16 and M-17), total soil (Tables M-18 through M-23), groundwater (Tables M-24 through M-29), and surface water (Table M-30). Risk estimates are compiled for all receptors by media in Table M-31.

Groundskeeper

Current Site Use. The groundskeeper is assumed to be exposed to surface soil outside the landfill boundary under the current site-use scenario. The total cancer risk of 2.07E-5 (Table M-31) exceeds the Florida point of departure of 1E-6 due to Aroclor 1260 and several PAHs (Table M-16). The total HI of 8.86E-2 falls below the threshold level of 1. Cancer-based RGOs for Aroclor 1260 and the PAHs in surface soil are presented in Table M-32.

Future Site Use. The groundskeeper is assumed to be exposed to total soil inside and outside the landfill boundary under the future site-use scenario. He is also assumed to be exposed to groundwater developed as a source of potable water. The total cancer risk of 4.51E-5 (Table M-31) exceeds the Florida point of departure of 1E-6, due largely to PCBs and several PAHs in total soil (Table M-18), and bis(2-ethylhexyl)phthalate and vinyl chloride in groundwater (Table M-24). The total HI of 9.81E-1 falls below the threshold level of 1. Cancer-based RGOs for COCs in total soil and groundwater are presented in Tables M-33 and M-34, respectively.

Construction Worker

Future Site Use. The construction worker is evaluated under the future site-use scenario only, where he is assumed to be exposed to total soil inside and outside the landfill boundary. He is also assumed to be exposed to groundwater by direct contact during construction projects, and also to groundwater developed as a source of potable water. The total cancer risk of 9.33E-7 (Table M-31) falls below the Florida point of departure of 1E-6. The total HI of 1 (rounded to

one significant figure) does not exceed the threshold level of 1. RGOs are not developed for the construction worker scenario.

Trespasser or Site Visitor

Current Site Use. The trespasser is assumed to be exposed to surface soil inside and outside the landfill boundary, surface water, and sediment under the current site-use scenario. The total cancer risk of 6.66E-7 (Table M-31) falls below the Florida point of departure of 1E-6. The total HI of 1.77E-2 falls below the threshold level of 1. RGOs are not developed for the trespasser under the current site-use scenario.

Future Site Use. The site visitor is assumed to be exposed to total soil inside and outside the landfill boundary, surface water, and sediment under the future site-use scenario. The total cancer risk of 1E-6 (rounded to one significant figure) (Table M-31) does not exceed the Florida point of departure of 1E-6. The total HI of 1.66E-2 falls below the threshold level of 1. RGOs are not developed for the site visitor under the future site-use scenario.

Resident

Future Site Use. The resident is evaluated under the future site-use scenario only, where he is assumed to be exposed to total soil inside and outside the landfill boundary, and groundwater developed as a potable source. The resident is also assumed to be exposed to surface water and sediment in the same manner and to the same extent as quantified for the site visitor under the future site-use scenario. Therefore, the risk estimates obtained for the site visitor are included in the risk sums for the resident. The total cancer risk of 1.53E-4 (Table M-31) exceeds the Florida point of departure of 1E-6, due largely to PCBs, dieldrin, and several PAHs in total soil (Table M-21), and bis(2-ethylhexyl)phthalate and vinyl chloride in groundwater (Table M-27). Cancer-based RGOs for COCs in total soil and groundwater are presented in Tables M-35 and M-36, respectively.

The total HI was estimated as 2.45E+0 for the adult resident and 7.05E+0 for the child resident (Table M-31), establishing the child evaluation as the more conservative. Therefore, discussion of noncancer effects is limited to the child resident. The total HI of 7.05E+0 (Table M-31) is due largely to antimony, PCBs, and TRPH in total soil (Table M-23), and to manganese, chlorobenzene and TRPH in groundwater (Table M-29). Noncancer-based RGOs for COCs in

total soil and groundwater for the child resident are presented in Tables M-37 and M-38, respectively.

5.9.1.3 Conclusions

The foregoing establishes soil and groundwater as the only media of concern; levels of TRPH in surface water contributed insignificantly to risk, and no COPCs were identified in sediment.

COCs in soil include Aroclor 1254, Aroclor 1260, dieldrin, and several PAHs. One Aroclor 1254 detection, four Aroclor 1260 detections, one dieldrin detection, and several PAH detections exceeded their respective SCTLs. Most SCTL exceedances occurred in surface soil. No pattern of distribution was apparent.

COCs in groundwater include manganese, bis(2-ethylhexyl)phthalate, chlorobenzene, vinyl chloride, and TRPH. Seven manganese detections exceeded the CTL, but only two exceeded the BSC. Concentrations in filtered and unfiltered samples were remarkably similar, suggesting that sediment contamination did not increase manganese concentrations in unfiltered samples. Among organics, bis(2-ethylhexyl)phthalate exceeded its CTL in four of the eight samples, vinyl chloride exceeded its CTL in two of three samples, and TRPH greatly exceeded its CTL in all three samples in which it was identified. The identification of a fairly large number of samples with bis(2-ethylhexyl)phthalate but no detections of TCE, and only one detection of chlorobenzene suggests that the vinyl chloride is present as a result of degradation of plastics rather than chlorinated solvents in the landfill.

The COCs identified, the relatively high concentrations of some of them (e.g., TRPH in groundwater), the lack of a pattern to their distribution, and the similarity between filtered and unfiltered samples suggests that the chemicals are site-related and probably reflect the contents of the landfill.

5.9.2 Landfill Area 3

5.9.2.1 Assessment of Contamination

Surface soil, total soil, groundwater, surface water, and sediment samples taken from Landfill Area 3 are identified in Tables M-39, M-40, M-41, M-42, and M-43, respectively. All media were analyzed for metals, PCBs, organochlorine pesticides, VOCs, SVOCs, and TRPH, although not each sample was analyzed for all chemical classes.

A total of 20 surface soil samples were taken – 12 from within the boundaries of the landfill and 8 from outside the boundaries of the landfill (Table M-39). COPCs identified in surface soil inside and outside the boundary of the landfill proper include the metals copper and mercury and low concentrations of two PAHs (Table M-44). The PAH benzo(a)pyrene was the only COPC selected in surface soil outside the boundary of the landfill (Table M-45). A total of eight samples of subsurface soil were taken – all from outside the boundaries of the landfill (Table M-40). COPCs in total soil were the same as those identified in surface soil inside and outside the boundary of the landfill proper (Table M-46). Higher concentrations of all COPCs were identified in the surface soil.

A total of 44 groundwater samples were taken and analyzed as described above (Table M-41). COPCs identified in groundwater include two metals (iron, manganese), TRPH, and several VOCs (Table M-47). Although the VOCs were identified in relatively few samples, concentrations were unusually high.

Four surface water and sediment samples were taken and analyzed as described above (Tables M-42 and M-43). Chemical detections in surface water were limited to metals, two SVOCs and TRPH (Table M-47). TRPH was selected as the only COPC in surface water (Table M-48). Chemicals detected in sediment include metals, PCBs, several PAHs, VOCs, and TRPH (Table M-49). COPCs were limited to two metals (arsenic, iron) and benzo(a)pyrene.

5.9.2.2 Risk Assessment and RGO Development

The same receptor scenarios and exposure pathways evaluated for Landfill Area 1 were evaluated for Landfill Area 3 (Table M-1). Risk estimates for each COPC, and risk estimates summed across all COPCs for a given medium, are compiled for surface soil (Tables M-50 and M-51), total soil (Tables M-52 through M-57), groundwater (Tables M-58 through M-63), surface water (Table M-64), and sediment (Table M-65). Risk estimates are compiled for all receptors by media in Table M-66.

Groundskeeper

Current Site Use. The groundskeeper is assumed to be exposed to surface soil outside the landfill boundary under the current site-use scenario. The total cancer risk of 9.01E-7 (Table M-66) falls below the Florida point of departure of 1E-6. An HI was not estimated for groundskeeper exposure to surface soil because no chemicals were selected as COPCs based on

noncancer effects. RGOs are not developed for the groundskeeper under the current site-use assumption.

Future Site Use. The groundskeeper is assumed to be exposed to total soil inside and outside the landfill boundary under the future site-use scenario. He is also assumed to be exposed to groundwater developed as a source of potable water. The total cancer risk of 8.85E-4 (Table M-66) exceeds the Florida point of departure of 1E-6, due to PAHs in total soil (Table M-52), and TCE and vinyl chloride in groundwater (Table M-58). The total HI of 1.65E+2 exceeds the threshold level of 1, due almost entirely to TCE, with significant contributions also from manganese, cis- and trans-1,2-DCE, vinyl chloride, and TRPH (Table M-58). RGOs for COCs in total soil and groundwater are presented in Tables M-67 and M-68, respectively.

Construction Worker

Future Site Use. The construction worker is evaluated under the future site-use scenario only, where he is assumed to be exposed to total soil inside and outside the landfill boundary. He is also assumed to be exposed to groundwater by direct contact during construction projects, and also to groundwater developed as a source of potable water. The total cancer risk of 1.83E-5 (Table M-66) exceeds the Florida point of departure of 1E-6, due largely to groundwater used as a potable source (Table M-59). The total HI of 1.79E+2 exceeds the threshold level of 1, due almost entirely to trichloroethylene in groundwater, with significant contributions also from manganese, cis- and trans-1,2-DCE, vinyl chloride, and TRPH (Tables M-59 and M-60). RGOs for COCs in groundwater developed as a potable source, and groundwater incidentally contacted during construction projects are presented in Tables M-69 and M-70, respectively.

Trespasser or Site Visitor

Current Site Use. The trespasser is assumed to be exposed to surface soil inside and outside the landfill boundary, surface water, and sediment under the current site-use scenario. The total cancer risk of 1.75E-7 (Table M-66) falls below the Florida point of departure of 1E-6. The total HI of 1.73E-3 falls below the threshold level of 1. RGOs are not developed for the trespasser under the current site-use scenario.

Future Site Use. The site visitor is assumed to be exposed to total soil inside and outside the landfill boundary, surface water, and sediment under the future site-use scenario. The total cancer risk of 1.65E-7 (Table M-66) falls below the Florida point of departure of 1E-6. The total

HI of 1.45E-3 falls below the threshold level of 1. RGOs are not developed for the site visitor under the future site-use scenario.

Resident

Future Site Use. The resident is evaluated under the future site-use scenario only, where he is assumed to be exposed to total soil inside and outside the landfill boundary, and groundwater developed as a potable source. The resident is also assumed to be exposed to surface water and sediment in the same manner and to the same extent as quantified for the site visitor under the future site-use scenario. Therefore, the risk estimates obtained for the site visitor are included in the risk sums for the resident. The total cancer risk of 5.21E-3 (Table M-66) exceeds the Florida point of departure of 1E-6, due largely to benzo(a)pyrene and dibenz(a,h)anthracene in total soil (Table M-55), and TCE and vinyl chloride in groundwater (Table M-61). Cancer-based RGOs for COCs in total soil and groundwater are presented in Tables M-71 and M-72, respectively.

The total HI was estimated at 4.72E+2 for the adult resident and 1.07E+3 for the child resident (Table M-66), establishing the child evaluation as the more conservative. Therefore, discussion of noncancer effects is limited to the child resident. The total HI of 1.07E+3 (Table M-66) is due almost entirely to TCE and cis-1,2-DCE, with significant contributions also from manganese, 1,2,4-trimethylbenzene, acetone, trans-1,2-DCE, vinyl chloride, and TRPH, all in groundwater (Table M-63). Noncancer-based RGOs for COCs in groundwater for the child resident are presented in Table M-73.

5.9.2.3 Conclusions

The foregoing establishes soil and groundwater as the only media of concern. Levels of TRPH in surface water, and metal and benzo(a)pyrene in sediment contributed insignificantly to risk.

COCs in soil are limited to two PAHs, benzo(a)anthracene and dibenz(a,h)anthracene. All detections – three benzo(a)pyrene and one dibenz(a,h)anthracene in surface soil – exceeded their SCTLs. No pattern of distribution was apparent.

COCs in groundwater include manganese, a wide variety of VOCs (acetone, 1,2,4-trimethylbenzene, cis- and trans-1,2-DCE, TCE, vinyl chloride) and TRPH. Eight of ten manganese detections exceeded the CTL, but only four exceeded the BSC. Concentrations in filtered and unfiltered samples were similar, suggesting that sediment contamination did not increase manganese concentrations in unfiltered samples. Acetone detections appeared to be

randomly distributed. TRPH was detected in only one shallow groundwater well (W-5-86) near the center of the landfill. This was unexpected, given its widespread distribution in soil. The chlorinated solvents were clearly concentrated at PZ02/MW06, suggesting that a release of solvents may have occurred near this location.

The COCs identified, the relatively high concentrations of some of them (e.g., solvents and TRPH in groundwater), the pattern of solvent distribution, and the similarity between filtered and unfiltered samples suggests that the chemicals are site-related and probably reflect the contents of the landfill.

5.10 Uncertainty Assessment

This section briefly discusses some of the sources of uncertainty that impact the LFNAS BHHRA. Generally, risk assessments carry two types of uncertainty. Measurement uncertainty refers to the usual variance that accompanies scientific measurements, e.g., instrument uncertainty (accuracy and precision) associated with determining contaminant concentrations. The results of the risk assessment reflect the accumulated variances of the individual measured values used to develop it. A different kind of uncertainty stems from data gaps, i.e., additional information needed to complete the database for the assessment.

Many sources of uncertainty are common to most risk assessments. These common sources of uncertainty are addressed by making conservative assumptions regarding EPC estimation, exposure assessment and toxicity assessment that impart a conservative bias to the risk assessment. It is beyond the scope of this document and quite unnecessary to identify and discuss each of these, because the measures taken collectively ensure protectiveness. There are, however, sources of uncertainty that are unique to the landfills BHHRA. They are discussed under the individual steps of the risk assessment as follows.

5.10.1 Data Evaluation

Probably the most important source of uncertainty in the data evaluation step is the potential for undetected chemicals to be present at levels that could contribute significantly to risk. This potential can be assessed by compiling the chemicals that were undetected in a given medium and comparing their Florida CTLs with their MDLs. Undetected chemicals whose CTLs fall below their MDLs may be present at concentrations that contribute significantly to risk. MDLs were not available in the data sets provided for the risk assessment; therefore, minimum and maximum RLs are presented in lieu of MDLs in the following tables. MDLs, however, are typically 2- to 5-fold or as much as an order-of-magnitude lower than their respective RLs.

Therefore, a reasonable range for the MDL can be estimated by dividing the RL by 2 and 5. The term, "MDL range," in the following discussions refers to the range of values obtained by dividing the RL by 2 and 5. The assessment for each former landfill and each medium follows.

Landfill Area 1

Soil. The assessment of nondetects in soil is presented in Table M-74 and discussed by chemical class.

Metals. The only metal whose CTL falls below its RLs is thallium, a naturally occurring metal that is typically present at low levels in all soils. Although the CTL falls below the maximum MDL range, the CTL exceeds the minimum MDL range. It is concluded that potentially undetected thallium in soil contributes minimally to the total uncertainty of the risk assessment.

PCBs and Pesticides. The CTLs for several undetected Aroclors fall below their respective RLs; however, the CTLs exceed their respective MDL ranges. It is concluded that toxicologically significant PCB levels below their RLs would have been reported as "J" qualified values and do not represent a source of uncertainty. The CTL for toxaphene falls below its maximum MDL range but above its minimum RL; potentially undetected toxaphene contributes minimally to total uncertainty.

SVOCs. Several undetected SVOCs have no CTLs; it is unclear the extent to which the potential presence of these chemicals in soil may contribute to total uncertainty. The CTLs for several SVOCs fall below their minimum or maximum RLs. Several CTLs also fall below their maximum MDL ranges and contribute to the uncertainty of the risk assessment. However, none of the CTLs fall below their minimum MDL ranges, and it is concluded that undetected SVOCs as a class contribute minimal uncertainty to the risk assessment.

VOCs. Several undetected VOCs have no CTLs; it is unclear the extent to which the potential presence of these chemicals may contribute to total uncertainty. The RLs for the remaining undetected VOCs, however, fall below their respective CTLs. The potential presence of these chemicals is judged not to contribute uncertainty to the risk assessment.

Groundwater

The assessment of nondetects in groundwater is presented in Table M-75 and discussed by chemical class.

Metals. The only metal whose CTL falls below its MDL range is thallium; this presents minimal uncertainty to the risk assessment.

PCBs and Pesticides. The CTLs for three undetected pesticides fall below their respective MDL ranges, which imparts a small amount to total uncertainty.

SVOCs. Several undetected SVOCs have no CTLs. The CTLs for several SVOCs fall below their minimum or maximum RLs and MDL ranges. Noteworthy in this group are several PAHs. The PAHs, however, have sufficiently high affinity for soil constituents that they are unlikely to leach to groundwater. Their potential presence represents a minor source of uncertainty.

VOCs. Several undetected VOCs have no CTLs; the potential presence of these chemicals contributes to the uncertainty of the risk assessment. The CTLs for several VOCs fall below their minimum or maximum RLs and MDL ranges and also contribute to the uncertainty. Noteworthy in this group are several halogenated alkanes and alkenes.

Surface Water

The assessment of nondetects in surface water is presented in Table M-76 and discussed by chemical class. It should be noted that groundwater CTLs based on drinking water use were used for the surface water screening, but that exposure to surface water is far less intensive than exposure to drinking water. This ameliorates the extent to which uncertainty from undetected chemicals in surface water reduces confidence in the risk assessment.

Metals. No metal CTLs fall below their respective RLs; undetected metals in surface water at Landfill Area 1 do not contribute uncertainty to the risk assessment.

PCBs and Pesticides. The CTLs for several PCBs and pesticides fall below their RLs; however, the CTLs fall below their MDL ranges only for aldrin, alpha-betahexachlorocyclohexane (BHC) and dieldrin. This is judged to contribute minimal uncertainty because exposure to surface water is far less than exposure to drinking water.

SVOCs. Several undetected SVOCs have no CTLs; it is unclear the extent to which the potential presence of these chemicals may contribute to total uncertainty. The CTLs for several SVOCs fall below their minimum or maximum RLs. Several CTLs also fall below their maximum MDL ranges. This is judged to contribute minimal uncertainty because exposure to surface water is far less than exposure to drinking water.

VOCs. Several undetected VOCs have no CTLs; it is unclear the extent to which the potential presence of these chemicals may contribute to total uncertainty. The CTLs for several VOCs fall below their minimum or maximum RLs. Several CTLs also fall below their maximum MDL ranges. With the possible exception of 1,2-dibromoethane, this is judged to contribute minimal uncertainty because exposure to surface water is far less than exposure to drinking water.

Sediment

The assessment of nondetects in sediment is presented in Table M-77 and discussed by chemical class. It should be noted that residential soil CTLs were used for the sediment screening, but that exposure to sediment is far less intensive than exposure to soil. This ameliorates the extent to which this source of uncertainty reduces confidence in the risk assessment.

Metals. The only metal whose CTL falls below its RLs is thallium; however, the CTL falls within the maximum MDL range. The potential for undetected thallium contributes nothing to the uncertainty.

PCBs and Pesticides. None of the CTLs for undetected PCBs or pesticides fall below their respective RLs. Undetected PCBs and pesticides contribute nothing to the uncertainty.

SVOCs. Several undetected SVOCs have no CTLs; it is unclear the extent to which the potential presence of these chemicals may contribute to total uncertainty. The CTLs for several SVOCs fall below their minimum or maximum RLs. Several CTLs also fall below their maximum MDL ranges and contribute to the uncertainty of the risk assessment. However, with the exception of N-nitroso-di-n-propylamine, none of the CTLs fall below their minimum MDL ranges.

VOCs. Several undetected VOCs have no CTLs; it is unclear the extent to which the potential presence of these chemicals may contribute to total uncertainty. The MDL ranges for the remaining undetected VOCs, however, fall below their respective CTLs.

Given the lesser exposure to sediment compared with soil, undetected chemicals in sediment are judged to contribute minimally to the total uncertainty of the risk assessment.

Landfill Area 3

Soil. The assessment of nondetects in soil is presented in Table M-78 and discussed by chemical class.

Metals. The only metal whose CTL falls below its RLs is thallium; however, the CTL falls within the maximum MDL range. It is concluded that potentially undetected thallium in soil contributes minimally to the total uncertainty of the risk assessment.

PCBs and Pesticides. The MDL range falls below the CTLs for all PCBs and pesticides; therefore, undetected PCBs and pesticides are considered not to contribute to total uncertainty.

SVOCs. Several undetected SVOCs have no CTLs; it is unclear the extent to which the potential presence of these chemicals in soil may contribute to total uncertainty. The CTLs for several SVOCs fall below their minimum or maximum RLs; however, the MDL ranges for these chemicals fall below their respective CTLs, and it is concluded that undetected SVOCs as a class contribute minimal uncertainty to the risk assessment.

VOCs. Several undetected VOCs have no CTLs; it is unclear the extent to which the potential presence of these chemicals may contribute to total uncertainty. The RLs for the remaining undetected VOCs, however, fall below their respective CTLs. The potential presence of these chemicals is judged not to contribute uncertainty to the risk assessment.

Groundwater

The assessment of nondetects in groundwater is presented in Table M-79 and discussed by chemical class.

Metals. No metal CTL falls below its RL; undetected metals in groundwater present no uncertainty to the risk assessment.

PCBs and Pesticides. The CTLs for several undetected pesticides fall below their respective MDL ranges, which contributes a small amount to total uncertainty.

SVOCs. Several undetected SVOCs have no CTLs. The CTLs for several SVOCs fall below their minimum or maximum RLs and MDL ranges and contribute to the uncertainty of the risk assessment. Noteworthy in this group are several PAHs, the dinitrotoluenes, and pentachlorophenol. The PAHs, however, are unlikely to leach to groundwater as explained above.

VOCs. Several undetected VOCs have no CTLs. The maximum RLs and MDL ranges for most of the VOCs in Landfill Area 3 groundwater exceed their respective CTLs because the RLs are unusually high. The high RLs for VOCs in general probably reflects the high levels of chlorinated solvents identified in MW06 and PZ02. The CTLs for several VOCs also fall below their minimum MDL ranges and contribute to the uncertainty of the risk assessment. Noteworthy in this group are several halogenated alkanes and alkenes.

Surface Water

The assessment of nondetects in surface water is presented in Table M-80 and discussed by chemical class. It should be noted that the application of groundwater CTLs based on drinking water use ameliorates the extent to which this source of uncertainty reduces confidence in the risk assessment.

Metals. No metal CTLs fall below their respective RLs; undetected metals in surface water at Landfill Area 3 do not contribute uncertainty to the risk assessment.

PCBs and Pesticides. The CTLs for several PCBs and pesticides fall below their RLs; however, the CTLs fall below their MDL ranges only for aldrin, alpha-BHC and dieldrin. This is judged to contribute minimal uncertainty because exposure to surface water is far less than exposure to drinking water.

SVOCs. Several undetected SVOCs have no CTLs; it is unclear the extent to which the potential presence of these chemicals may contribute to total uncertainty. The CTLs for several SVOCs fall below their minimum or maximum RLs. Several CTLs also fall below their maximum MDL ranges. This is judged to contribute minimal uncertainty because exposure to surface water is far less than exposure to drinking water.

VOCs. Several undetected VOCs have no CTLs; it is unclear the extent to which the potential presence of these chemicals may contribute to total uncertainty. The CTLs for several VOCs fall below their minimum or maximum RLs. Several CTLs also fall below their maximum MDL ranges. With the possible exception of 1,2-dibromoethane, bromomethane and isopropylbenzene, this is judged to contribute minimal uncertainty because exposure to surface water is far less than exposure to drinking water.

Sediment

The assessment of nondetects in sediment is presented in Table M-81 and discussed by chemical class.

Metals. The only metal whose CTL falls below its RLs is thallium. However, the CTL falls within the maximum MDL range.

PCBs and Pesticides. None of the CTLs for undetected PCBs or pesticides fall below their respective RLs.

SVOCs. Several undetected SVOCs have no CTLs; it is unclear the extent to which the potential presence of these chemicals in sediment may contribute to total uncertainty. The CTLs for several SVOCs fall below their minimum or maximum RLs, but not below their maximum MDL ranges, with the exception of N-nitroso-di-n-propylamine.

VOCs. Several undetected VOCs have no CTLs; it is unclear the extent to which the potential presence of these chemicals may contribute to total uncertainty. The MDL ranges for the remaining undetected VOCs, however, fall below their respective CTLs.

Given the lesser exposure to sediment compared with soil, undetected chemicals in sediment are judged to contribute minimally to the total uncertainty of the risk assessment.

5.10.2 Exposure Assessment

The landfills currently are unused and there are no plans to develop these areas for use in the future. Both landfills, however, reside within the Reynolds Industrial Park, which suggests that commercial or industrial development in the future is plausible. Residential development,

however, is very unlikely. Therefore, inclusion of an on-site resident, generally considered the most conservative exposure scenario, imparts a conservative bias to the risk assessment.

Similarly, shallow groundwater is currently unused and there are no plans to use the groundwater in the future. Contact with shallow groundwater during construction projects is a plausible exposure scenario. Development of groundwater as a potable source, however, is very unlikely. Therefore, evaluation of shallow groundwater as a source of potable water imparts a highly conservative bias to the risk assessment.

5.10.3 Toxicity Assessment

The decision to exclude iron in groundwater at Landfill Area 1, and iron in groundwater and sediment at Landfill Area 3 from quantitative evaluation may contribute a non-conservative source of uncertainty to the assessment. Daily estimates of iron intake from these sources range from 5 mg/day for the trespasser or site visitor exposed to sediment at Landfill Area 3, to 46 mg/day for adult residents exposed to groundwater at Landfill Area 1 (data not shown). These intakes are below the range of 50 to 100 mg/day associated with chronic iron overload in Bantu consuming Kafir beer brewed in iron vessels (EPA, 1996b). It was determined that the iron in the Kafir beer was unusually bioavailable because of low pH and the presence of alcohol. Also, the 5 to 46 mg/day intake range is below the dietary range of 98 to 1418 mg/day estimated for Ethiopians, who reportedly have the highest natural dietary iron intake (EPA, 1996b). Iron overload has not been observed in the Ethiopians. It is concluded that exclusion of iron from the quantitative assessment is unlikely to contribute a non-conservative bias to the risk assessment.

6.0 Screening-Level Ecological Risk Assessment

Site-specific screening-level ecological risk assessments (SLERA) are presented for Landfill Areas 1 and 3 at the LFNAS in Green Cove Springs, Florida to determine if DOD-related chemicals present at the landfills have the potential to pose unacceptable risk to ecological receptors. The goal of a SLERA is to conservatively assess ecological risk potential via a screening-level effect evaluation and preliminary problem formulation. The screening-level effects evaluation compares site-related chemical concentrations to conservative benchmark values. The preliminary problem formulation expands upon the results of the screening-level effect evaluation by presenting an ecologically-based conceptual model for the landfills. This conceptual model addresses the environmental setting, chemicals known or suspected of being present based on historical use, presence or absence of contaminant transport mechanisms, and presence or absence of viable habitat, receptors, and exposure pathways. In addition, general modes of toxicity and bioaccumulation tendencies of any identified chemicals are evaluated.

These SLERAs are conducted in general accordance with the EPA guidance document *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments* (EPA, 1997d), also known as the Process Document. The SLERAs also include recent modifications in the ecological risk assessment procedure as described in the memorandum *Amended Guidance on Ecological Risk Assessment at Military Bases: Process Considerations, Timing of Activities, and Inclusion of Stakeholders* (EPA, 2000a) as well as *ECO Update Number 12: The Role of Screening-Level Risk Assessments and Refining Contaminants of Concern in Baseline Risk Assessments* (EPA, 2001b). The revised approach essentially combines Steps 2 and 3a of the eight-step process presented in the Process Document (Figure 6-1), such that screening-level problem formulation (Step 3a) is developed and presented with the effect evaluation. The methods and procedures used in this SLERA follow those described in the work plan (IT, 2000b).

General site descriptions and history of the two landfills, as well as previous studies, are presented in Chapter 3.0. Section 6.1 presents an overview of site ecology, including habitat type, threatened and endangered (T&E) species, and exposure pathways. Section 6.2 presents a description and review of analytical data used in the SLERAs. The SLERA process conservatively assesses ecological risk potential by comparing contaminant levels to chemical-specific toxicity benchmark values in the screening-level effect evaluation, which is presented in Section 6.3, along with uncertainties associated with the evaluations. Section 6.4 presents the preliminary problem formulation, and includes a comparison of inorganic analytes to background

levels, a spatial analysis of chemicals that exceed screening criteria, conceptual site exposure models, and an evaluation of the toxicity of site-related chemicals. Section 6.5 identifies chemicals that warrant further ecological investigation, and Section 6.6 presents recommendations based on the findings.

6.1 Ecological Habitat and Receptors

Although much of the LFNAS is covered with runways and light industrial facilities, there does exist a significant amount of excellent habitat on and adjacent to the site that likely supports a diverse array of wildlife. As observed during the site walk conducted in February 2000, much of the area surrounding LFNAS is considered to be of high ecological value (IT, 2000b). Approximately 18,500 acres of state-managed land is located to the south and east as the Bayard Conservation Area (Preservation 2000, 1997). Additionally, to the southeast lies an area identified as potential habitat for rare species (Florida Natural Areas Inventory [FNAI], 2001).

6.1.1 Habitat Types

The LFNAS is located along the southern bank of the St. Johns River. Much of the LFNAS was built on sandy fill material dredged locally from Red Bay. Forested hardwood wetlands exist immediately to the south and southwest of the airstrips. An area of upland pine forest is located on the western side as well as the southeast corner of the site. The Bayard Conservation Area, managed by the SJRWMD as part of the St. Johns River Wildlife Management Area, lies to the east of LFNAS. This preserve consists of 18,500 acres of pine flatwoods, sandhills, bottomland hardwoods, and 10 miles of river frontage (Preservation 2000, 1997). A golf course is located adjacent to the eastern boundary of the LFNAS in the north. Figure 6-2 depicts the general habitat within and immediately adjacent to LFNAS.

Landfill Area 1. The dominant habitat type at Landfill Area 1 is shrub/scrub. The landfill is distinctly higher in elevation than the surrounding terrain. A few deciduous and coniferous pine trees are located on the western side of the landfill. To the east, outside of the Reynolds Industrial Park boundary, is the Bayard Conservation Area, which is characterized by open grassland. To the north, on the opposite side of Wildwood Road, is a large ditch in which alligators have reportedly been observed. Beyond the ditch is a dense mixed pine/hardwood forest. A second ditch is located immediately to the east of the landfill. During the February 2000 site visit, these ditches contained water, however, no flow was observed. A deciduous wetland forest lies to the south and east of Landfill Area 1, in which kingfishers and hooded mergansers were observed (IT, 2000c).

Landfill Area 3. Landfill Area 3 is dominated and surrounded by pine forest, with an average breast-height diameter of approximately 8 inches. Some areas of which are characterized by moderately dense understory brush, while other areas have no vegetative ground cover. Soil at the site is sandy, and a gopher tortoise borough was observed east of the site. A shallow pool of water, almost completely covered with duckweed, exists within the landfill area. A limited amount of wetland vegetation surrounds the pond. No fish were observed in the pond during the February 2000 site visit, which is most likely too small and shallow to support fish; however, seasonal benthic invertebrate species would not be unexpected. A narrow shallow ditch was noted on the western side of the pond, but it could not be determined if the ditch feeds or drains the pond, as there was no observed flow (IT, 2000c).

6.1.2 Receptors of Special Status

No LFNAS-specific T&E species surveys could be located. However, inquires to the Florida Fish and Wildlife Conservation Commission (FFWCC) and the FNAI indicate species of special concern may be present on or in close proximity (i.e., within 3 miles) to the LFNAS.

The FFWCC reported that although no listed species exist in their database for the LFNAS property, the indigo snake and gopher tortoise are likely to inhabit the area. Additionally, a bald eagle nest, active from 1996 to 2000, is located approximately 2.75 miles southeast of LFNAS at coordinates 29° 57.05' / 81° 37.42'. Appendix N presents the FFWCC reports.

The FNAI report (Appendix N) indicated several T&E species have occurred at the LFNAS, including osprey, painted bunting, and southeast American kestrel. Additionally, T&E bird species have been reported within three miles of the site, and include the bald eagle, osprey, painted bunting, swallow-tailed kite, and American redstart. The FNAI database includes a report of a gopher tortoise approximately 2.5 miles southwest of the site.

During the February 2000 site visit, which was not intended to be a formal T&E species survey, the Shaw biologist observed an abandoned gopher tortoise borough east of Landfill Area 3. It is important to note that the sandy fill material used to build the LFNAS provides excellent gopher tortoise borough substrate. A live gopher tortoise was observed at the pyridine/burn area during the February 2000 site visit.

6.1.3 Potential Exposure Pathways

Exposure pathways describe the path that a chemical may take from a source to potential receptors. Figures 6-3 and 6-4 pictorially present exposure routes at Landfill Areas 1 and 3, respectively.

Landfill Area 1. The primary potential pathways for contaminant migration include groundwater percolation and overland runoff to adjacent soil, the ditches to the north and west, as well as the wetland to the south of the site. Therefore, this SLERA investigates potential risk to both terrestrial and aquatic receptors.

Landfill Area 3. The primary potential pathways for contaminant migration include groundwater percolation and overland runoff to adjacent soil, as well as to the small water body within the site. Therefore, this SLERA investigates potential risk to both terrestrial and aquatic receptors.

Exposure pathways from the media at both landfills to receptors may include the following routes or mechanisms:

- Soil
  - Direct (passive) uptake
  - Ingestion of contaminated food items
  - Incidental ingestion of soil
  - Dermal contact
  - Inhalation of vapors or dust.

- Surface Water
  - Direct (passive) uptake
  - Ingestion of water
  - Ingestion of contaminated food items
  - Dermal contact
  - Inhalation of vapors.

- Sediment
  - Direct (passive) uptake
  - Ingestion of contaminated food items
  - Incidental ingestion of sediment
  - Dermal contact.

6.2 Review, Evaluation, and Presentation of Analytical Data

Data collected during the October 2000 and April 2001 field efforts are evaluated for their potential threat to ecological receptors in the SLERAs for each landfill. Detections that are

below the RL, but above the MDL (i.e., "J" qualified) are considered estimates and are treated as non-qualified data in the SLERAs.

Data for landfills are evaluated separately for each medium. Data for both sites include surface soil, surface water, and sediment. For ecological impacts, soil collected from 0 to 1 foot bgs is considered surface soil.

6.2.1 Landfill Area 1 Analytical Data

For ecological purposes, surface soil, surface water and sediment were collected at Landfill Area 1 (Figure 1-3). Table 6-1 presents samples used for the Landfill Area 1 SLERA.

Surface Soil. Twenty locations were sampled. All samples were analyzed for TAL metals, VOCs, SVOCs, pesticides, PCBs, and TRPH. Additionally, ten were analyzed for TOC and three samples were analyzed for PAHs. As TRPH consists of a "cocktail" of organic compounds, it will not be included in the SLERA evaluation. Rather individual compounds will be evaluated (e.g., VOCs and SVOCs). TOC was collected for comparative purposes and is not assessed in the SLERA.

Surface Water. Four surface water locations were sampled. All were analyzed for TAL metals, VOCs, SVOCs, pesticides, PCBs, TRPH, alkalinity, and hardness. Additionally, one of the samples was analyzed for PAHs. As TRPH consists of a "cocktail" of organic compounds, it will not be included in the SLERA evaluation. Rather individual compounds will be evaluated (e.g., VOCs and SVOCs). Alkalinity and hardness were collected for comparative purposes and are not assessed in the SLERA.

Sediment. Four sediment locations, collocated with the surface water locations, were sampled. All were analyzed for TAL metals, VOCs, SVOCs, pesticides, PCBs, TRPH, and TOC. Additionally, one of the samples was analyzed for PAHs. As TRPH consists of a "cocktail" of organic compounds, it will not be included in the SLERA evaluation. Rather individual compounds will be evaluated (e.g., VOCs and SVOCs). TOC was collected for comparative purposes and is not assessed in the SLERA.

6.2.2 Landfill Area 3 Analytical Data

For ecological purposes, surface soil, surface water and sediment were collected at Landfill Area 3 (Figure 1-4). Table 6-2 presents samples used for the Landfill Area 3 SLERA.

Surface Soil. Twenty locations were sampled. All samples were analyzed for TAL metals, VOCs, SVOCs, pesticides, PCBs, and TRPH. Additionally, twelve were analyzed for TOC and one sample was analyzed for PAHs. As TRPH consists of a "cocktail" of organic compounds, it will not be included in the SLERA evaluation. Rather individual compounds will be evaluated (e.g., VOCs and SVOCs). TOC was collected for comparative purposes and is not assessed in the SLERA.

Surface Water. Four surface water locations were sampled. All were analyzed for TAL metals, VOCs, SVOCs, pesticides, PCBs, TRPH, alkalinity, and hardness. Additionally, one of the samples was analyzed for PAHs. As TRPH consists of a "cocktail" of organic compounds, it will not be included in the SLERA evaluation. Rather individual compounds will be evaluated (e.g., VOCs and SVOCs). Alkalinity and hardness were collected for comparative purposes and are not assessed in the SLERA.

Sediment. Four sediment locations, collocated with the surface water locations, were sampled. All were analyzed for TAL metals, VOCs, SVOCs, pesticides, PCBs, TRPH, and TOC. Additionally, one sample was analyzed for PAHs. As TRPH consists of a "cocktail" of organic compounds, it will not be included in the SLERA evaluation. Rather individual compounds will be evaluated (e.g., VOCs and SVOCs). TOC was collected for comparative purposes and will not be assessed in the SLERA.

6.3 Screening-Level Effects Evaluation

The effects evaluation is to assesses the potential of a chemical stressor to be present at a level that adversely affects viable populations, communities, and/or ecosystems at, or near, a site. The first step of this process is to compare concentrations of chemicals detected on the site to conservative ecological screening values. This step is intended to develop a list of constituents that are of primary ecological concern for the site. Generic EPA Region 4 screening values, or ecological benchmark screening values (EBSV), are used as nonreceptor-specific benchmarks. Appendix N-2 presents EBSVs for soil, surface water, and sediment.

6.3.1 Preliminary Screening and Abiotic Hazard Quotients

Identification of chemicals of potential ecological concern (COPEC) consists of dividing the MDC of each chemical by its corresponding EBSV for the appropriate medium to arrive at a screening-level hazard quotient (HQ_{screen}), as follows:

$$HQ_{screen} = MDC / EBSV$$

The HQ_{screen} provides the risk assessor with a numerical value that is proportional to the screening-level hazard potential associated with each chemical in the corresponding medium. An HQ_{screen} value less than or equal to 1 indicates that the chemical's MDC is less than the conservative EBSV and, therefore, unlikely to pose a hazard to ecological receptors exposed to that medium. Conversely, HQ_{screen} values greater than 1 indicate that the chemical may have the potential to pose risk to receptors exposed to that medium. For some chemicals, EBSVs are not available. Therefore, these chemicals are identified as COPEC. Chemicals identified as COPEC are carried forward into the problem formulation step (Step 3a).

6.3.2 Identification of Chemicals of Potential Ecological Concern

COPEC may vary from site to site based on historical use and activities that have taken place. Summary tables for each medium at each site present the following information:

- Chemical name
- Frequency of detection
- Range of detected concentrations
- Range of reporting limits
- Mean chemical concentration
- Chemical-specific EPA Region 4 EBSV
- HQ_{screen} value
- COPEC identification conclusion: No (excluded) or Yes (selected)
- Basis for inclusion
- Number of samples that exceed the EBSV.

6.3.2.1 COPEC Summary for Landfill Area 1

Surface Soil. As presented in Table 6-3, fourteen inorganic analytes, four PAHs, six pesticides, and two PCBs were selected as COPECs because their MDCs exceeded their respective EBSVs. Three inorganic analytes and one VOC were identified as COPECs because no EBSVs were available.

Surface Water. As presented in Table 6-4, ten inorganic analytes were selected as COPECs because their MDCs exceeded their respective EBSVs.

Sediment. As presented in Table 6-5, four inorganic analytes, one pesticide, and two PCBs were selected as COPECs because their MDCs exceeded their respective EBSVs. Nine inorganic analytes were identified as COPECs because no EBSVs were available.

6.3.2.2 COPEC Summary for Landfill Area 3

Surface Soil. As presented in Table 6-6, eight inorganic analytes and five PAHs were selected as COPECs because their MDCs exceeded their respective EBSVs. Three inorganic analytes, one VOC, and two SVOCs were identified as COPECs because no EBSVs were available.

Surface Water. As presented in Table 6-7, eleven inorganic analytes and one SVOC were selected as COPECs because their MDCs exceeded their respective EBSVs. One VOC was selected as a COPEC because no EBSV was available.

Sediment. As presented in Table 6-8, one inorganic analyte, one VOC, four PAHs, one SVOC, and one PCB were selected as COPECs because their MDCs exceeded their respective EBSVs. Eleven inorganic analytes and one VOC were identified as COPECs because no EBSVs were available.

6.3.3 Uncertainty in the Screening-Level Effects Evaluation

As the SLERA is broad in scope and is designed as an initial step in the determination of whether site contaminants present potential risk to ecological receptors, there exist uncertainties that must be considered in the evaluation of risk.

By design, SLERAs have a high degree of uncertainty. The assumptions employed in a SLERA result in the "margin of error" being on the conservative or protective side of the risk scale. It is therefore safe to assume that chemical levels resulting in HQ_{screen} values less than 1 do not pose a significant risk to ecological receptors. Additionally, chemicals present at levels resulting in HQ_{screen} values greater than 1 may not necessarily pose risk to ecological receptors at the site, as EBSVs are intended to be protective of 95 percent of the floral and faunal species 95 percent of the time. The basis of the EBSVs used, presence of receptors, and likelihood of exposure are additional factors that need to be considered before ecological stress can be evaluated. These additional evaluations are required to reduce the uncertainty in the SLERA process.

Ideally, the sampling program should reflect selection of sample locations with a range of contaminant concentrations. In addition, the number of collected samples should be adequate to characterize the aerial extent of contamination. The MDC, as representative of the level of contamination at a site, may overestimate the threat posed to widely dispersed wildlife. To reduce this uncertainty, an evaluation of the number and location of samples exceeding EBSVs as well as background levels are necessary.

Certain inorganic analytes such as calcium, magnesium, potassium, and sodium are considered to be macronutrients essential to many organisms for normal healthy development. These analytes are not expected to produce adverse ecological effects unless the detected concentration substantially exceeds background levels.

Due to the conservative nature of the EBSVs, many are less than concentrations of naturally occurring metals and PAHs. COPECs at naturally occurring levels do not represent contribution from site-related activities. Inorganic and PAH COPEC exceeding both the EBSV and the background levels are considered to be of highest priority for closer evaluation at the site.

6.4 Preliminary Problem Formulation

EPA Region 4 allows for a preliminary problem formulation to be developed in conjunction with the screening-level effects analysis in order to refine the COPEC selection process and "present a reasoned toxicological approach for the elimination of one or more COPEC from future consideration" (EPA, 2000a). The problem formulation step of the risk assessment process addresses many of the uncertainties raised in the SLERA, and provides a "reality check" to the conservative screening level analysis.

Site-specific information on background, magnitude of screening-level exceedance, distribution of contaminants, and potential exposure to receptors are used to evaluate the potential for risks to ecological receptors and to more clearly define potential problems at each site. A comparison of concentrations of inorganic analytes and PAHs to background levels is provided in Section 6.4.1. Section 6.4.2 discusses the spatial distribution of COPECs at each site and Section 6.4.3 discusses conceptual site exposure, which describes receptors and exposure pathways that are likely to exist at the landfills. Section 6.4.4 discusses the basis of the EBSVs and their relevance, and general chemical characteristics are presented and discussed in Section 6.4.5.

6.4.1 Comparison to Background

An important point the comparison to EBSVs does not take into consideration is that the presence of certain classes of chemicals, such as metals and PAHs, may not be the result of historical activities at the landfill. Metals are naturally occurring, originating from their respective source rocks, and thus their presence in soil samples may not be indicative of contamination. Other compounds, such as PAHs, are formed primarily as the result of incomplete combustion, and include both anthropogenic (e.g., combustion of fossil fuels) and

natural sources (e.g., forest fires). PAHs tend to be widely dispersed in the environment as the result of non-point source contamination.

The purpose of this comparison is to determine whether the presence of inorganic analytes and PAHs are the result of past DOD activities at the site. All metals and PAHs that were identified as COPEC in Section 6.3.2 (Tables 6-3 through 6-8) were compared to BSC. For each detected chemical, the site MDC was compared to either the 95th UTL (for normally and lognormally distributed analytes) or the 95th UCL of the 95th percentile (for nonparametrically distributed analytes).

Analytes that exceed the BSC are not necessarily contaminants, but should be considered suspect and evaluated further. The site and background data sets were subjected to a more rigorous comparison to determine if the metals concentrations detected in soil represent naturally occurring concentrations (Appendices K and L; summarized in Section 4.3). The hot measurement test, in which the site MDC is compared to the BSC, was performed to detect potential hot spots (EPA, 1989c, 1992d, and 1994b). The WRS test, which tests for differences between the medians of two data sets, was also performed (EPA, 2000b). The WRS test is sensitive to slight to pervasive contamination. The hot measurement test and WRS test are thus complementary, because they test for different modes of contamination. Box-and-whisker plots were prepared to visually compare the site and background data sets and to properly interpret the results of the WRS test. Any analyte that failed one or both quantitative statistical tests was then subjected to geochemical evaluation, which is based on the natural associations of trace elements with specific minerals in the soil/sediment matrix, or with suspended particulates in surface water. Correlation plots of trace element versus major element concentrations were constructed to identify potentially contaminated samples.

6.4.1.1 Landfill Area 1 Background Comparison

Surface Soil. As presented in Table 6-9, sixteen of the 17 metals, initially identified as COPEC, are present at concentrations that exceed the BSC. Aluminum, however, is shown to be present at concentrations within the BSC. Furthermore, as discussed in Section 4.3, only antimony is present in certain samples at anomalously high concentrations relative to the major elements, and may contain a component of contamination. The remaining elements that failed statistical comparison to background exhibit only naturally occurring concentrations in the site samples (Appendix K-1).

All four PAHs (benzo[a]pyrene, fluoranthene, phenanthrene, and pyrene) initially identified as COPEC exceeded their BSC (Table 6-9), and may be the result of historical landfill activity or deposition from anthropogenic sources.

Surface Water. As presented in Table 6-10, chromium, lead, and thallium concentrations exceeded their respective BSC; however, these concentrations were shown to be naturally occurring (Appendix K-4). The remaining seven inorganic analytes are present at concentrations within range of background.

Sediment. As presented in Table 6-11, six (calcium, copper, lead, mercury, vanadium, and zinc) of the 13 inorganic analytes initially identified as COPECs are present at concentrations that exceed the BSC; however, as presented in Appendix K-3, the detected concentrations of these metals are within range of background. Detected concentrations of the remaining seven analytes (aluminum, barium, iron, magnesium, manganese, potassium, and sodium) were shown to be naturally occurring.

6.4.1.2 Landfill Area 3 Background Comparison

Surface Soil. As presented in Table 6-12, nine of the 11 metals, initially identified as COPEC, are present at concentrations that exceed the BSC. Aluminum and vanadium are present at concentrations less than the BSC, and as presented in Appendix L-1, vanadium is within range of background. Calcium, chromium, iron, potassium, and sodium are shown to be naturally occurring. However, concentrations of copper, lead, mercury, and zinc are anomalously high in certain samples and may contain a component of contamination.

Four of the five PAHs (benzo(a)pyrene, fluoranthene, phenanthrene, and pyrene) initially identified as COPEC exceeded their respective BSC (Table 6-12), and may be the result of historical landfill activity or deposition from anthropogenic sources. Anthracene did not exceed its BSC.

Surface Water. As presented in Table 6-13, chromium and lead concentrations exceeded their respective BSC; however, as presented in Appendix L-4, geochemical evaluation indicates that detected concentrations of these two metals are naturally occurring. The remaining nine inorganic analytes are at concentrations less than their respective BSC.

Sediment. As presented in Table 6-14, aluminum, barium, beryllium, calcium, magnesium, manganese, potassium, selenium, sodium, and vanadium are present at concentrations less than their respective BSC. Iron and mercury are present at concentrations that exceed their respective BSC; however, as presented in Appendix L-3, geochemical evaluation indicates that detected concentrations of mercury are naturally occurring. The concentration of iron is shown to be anomalously high at one sample location (SD04), and may contain a component of contamination.

Only one of four PAHs (benzo[a]pyrene) initially identified as COPEC exceeded the BSC (Table 6-14), and may be the result of historical landfill activity or deposition from anthropogenic sources. Concentrations of chrysene, fluoranthene, and pyrene, however, were less than their BSC and are considered to be naturally occurring.

6.4.2 Spatial Analysis of COPEC

The spatial distribution of COPECs that exceed their EBSVs across a site can aid in the identification of "hot spots" and also the extent that ecological receptors may be exposed. The following presents COPEC analytical results for each sample location and indicates whether the medium concentration exceeds the EBSV and/or background level.

6.4.2.1 Landfill Area 1 Spatial Analysis

Surface Soil. Chromium and zinc exceed both the EBSV and background screening criterion at the majority (13 of 20 and 15 of 20 samples, respectively) of the sample locations. Iron and lead are fairly prevalent (9 of 20 and 8 of 20 samples, respectively) at concentrations that exceed both the EBSV and BSC. The four PAH COPECs exceed both EBSVs and background criterion in five samples (LF1SS05, LF1SS06, LF01SB02, LF1SB04, and LF1SB06). Pesticide detections that exceeded EBSVs are concentrated at sample locations LF1SS01, LF1SS02, LF1SS03, and LF1SS05, all of which are located in the eastern portion of the landfill (Figure 1-3). PCBs, primarily Aroclor 1260, exceeded EBSVs in 15 of 20 samples.

Individual sample locations with the greatest number of COPEC exceedances include LF1SS03, LF1SB04, LF1SB05, and LF1SB06. Three of these samples (LF1SS03, LF1SB04, and LF1SB05) are located in close proximity to one another in the southeast corner of the site along the embankment adjacent to the cypress swamp (Figure 1-3).

Surface Water. Only three inorganic analytes (chromium, lead, and thallium) exceeded both EBSVs and BSCs. Chromium and thallium exceeded at LF1SW02, and lead exceeded at LF1SW04. Sample locations are presented in Figure 1-3.

Sediment. Only one sample location (LF1SD04) had concentrations of metals (copper, lead, mercury, and zinc) that exceeded EBSVs and BSCs as well as one pesticide (4,4'-DDE) and two PCBs (Aroclor 1254 and Aroclor 1260) that exceeded their respective EBSVs. Although no EBSVs were available for calcium and vanadium, the LF1SD04 concentrations exceeded their background criteria.

6.4.2.2 Landfill Area 3 Spatial Analysis

Surface Soil. Inorganic analytes exceeded their EBSVs and/or background screening criteria at all 20 samples collected on the site. Sample locations that have multiple metals and exceeded both EBSV and BSC include LF3SS03, LF3SS04, LF3SS05, LF03SS07, and LF3SS08. Three of these samples (LF3SS03, LF3SS07, and LF3SS08) are clustered in the central portion of the site (Figure 1-4).

SVOCs, primarily PAHs, are of most concern in samples LF3SS03, LF3SS11, and LF3SB05. Samples LF3SS11 and LF3SB05 are located on the eastern side of the site in close proximity to each other, while LF3SS02 is located on the opposite side of the landfill (Figure 1-4).

Surface Water. Chromium and lead were the only metals of concern based on comparison to EBSVs and BSCs. Chromium exceeded both EBSV and BSC at only one location (LF3SW02), and lead exceeded both EBSV and background criteria at three sample points (LF3SW01, LF3SW02, and LF3SW04) (Figure 1-4). bis(2-Ethylhexyl)phthalate exceeded its EBSV at LF3SW01, LF3SW02, and LF3SW04.

Sediment. Iron exceeded its BSC in LF3SD04, and mercury exceeded both the EBSV and BSC at LF3SD03. Acetone was detected in LF3SD04 at a concentration exceeding the EBSV, and LF3SD03 had concentrations of benzo(a)pyrene and Aroclor 1260 exceeding the EBSV and/or background. LF3SD01 and LF3SD02 had EBSV exceedances of benzoic acid and Aroclor 1260, respectively.

6.4.3 Conceptual Site Exposure Model

Exposure pathways describe the route that a chemical may take from a source to potential receptors.

6.4.3.1 Landfill Area 1 Conceptual Site Exposure Model

Figure 6-3 pictorially presents exposure routes at Landfill Area 1. The primary potential pathways for contaminant migration include groundwater percolation and overland runoff to adjacent soil and water bodies located immediately to the north and south of the landfill.

Exposure pathways from contaminated media at Landfill Area 1 to receptors may include the following routes or mechanisms:

- Direct (passive) uptake (e.g., root or foliar absorption)
- Ingestion of contaminated food items
- Ingestion of water
- Incidental ingestion of soil and/or sediment
- Dermal contact
- Inhalation of vapors or dust.

The primary route of exposure at Landfill Area 1 is for organisms that feed on plants that live in contaminated media and higher trophic level organisms that feed on plants or organisms that have accumulated chemicals in their tissues by feeding in contaminated areas.

6.4.3.2 Landfill Area 3 Conceptual Site Exposure Model

Figure 6-4 pictorially presents exposure routes at Landfill Area 3. The primary potential pathways for contaminant migration include groundwater percolation and overland runoff to adjacent soil and water bodies located within the landfill.

Exposure pathways from contaminated media at Landfill Area 3 to receptors may include the following routes or mechanisms:

- Direct (passive) uptake (e.g., root or foliar absorption)
- Ingestion of contaminated food items
- Ingestion of water
- Incidental ingestion of soil and/or sediment
- Dermal contact
- Inhalation of vapors or dust.

The primary route of exposure at Landfill Area 3 is for organisms that feed on plants that live in contaminated media and higher trophic level organisms that feed on plants or organisms that have accumulated chemicals in their tissues by feeding in contaminated areas.

6.4.4 Basis of Ecological Benchmark Screening Values

When considering potential ecological risk, it is important to understand the basis of the EBSV the site concentration is compared to, as well as alternate benchmarks that may be used in the evaluation.

6.4.4.1 Surface Soil

Antimony. The EPA Region 4 benchmark screening criteria of 3.5 mg/kg was derived by the Dutch National Institute of Public Health and the Environment (RIVM, 1997) based on background concentrations in the Netherlands and an ecological effect level to determine maximum permissible concentrations (MPC). Therefore, the uncertainty surrounding this criterion is significant, because background concentrations in the Netherlands have no connection to biological effects either at the site or in the Netherlands.

Efroymson et al. (1997a) derived an alternative benchmark of 5 ppm based on a report of unspecified toxic effects on plants grown in a surface soil with the addition of 5 ppm antimony (Kabata-Pendias and Pendias, 1984). Because of the paucity of details in this study, confidence in the alternative benchmark is low.

Copper. The EPA Region 4 benchmark screening criteria of 40 mg/kg was derived by the Dutch National Institute of Public Health and the Environment (RIVM, 1997) based on background concentrations in the Netherlands and an ecological effect level to determine the MPC. Therefore, the uncertainty surrounding this criterion is significant, because background concentrations in the Netherlands have no connection to biological effects either at the site or in the Netherlands.

An alternative benchmark of 100 ppm based on phytotoxicity effects was presented in Efroymson et al. (1997a). This benchmark was derived from a study conducted by Wallace et al. (1997) that evaluated the effects of Cu added as $CuSO_4$ to a loam soil. Leaf and stem weights of bush beans grown from seed for 17 days were reduced by 26 percent by 200 ppm soil; 100 ppm

had no effect. Confidence in the benchmark of 100 ppm is considered low because of the low number of studies (Efroymson et al., 1997a).

Efroymson (1997b) presents a second alternative benchmark of 50 ppm for copper based on earthworm effects. An adequate number of studies was available to derive a 10th percentile of toxicity effects; however, this value was presented as 60 ppm in the text summaries of the copper toxicity studies rather than 50 ppm as the appropriate benchmark to use. It is not known which value is correct. Confidence in the benchmark is moderate (Efroymson et al., 1997b).

A third alternative benchmark value of 100 ppm was presented in Efroymson (1997b), as well, based on toxic effects to soil microorganisms and microbial processes. Sufficient data were available to derive a 10th percentile, which was adopted as the benchmark value. Confidence in this benchmark is high (Efroymson et al., 1997b).

Lead. The EPA Region 4 benchmark screening criteria of 50 mg/kg is based on multiple studies, including an investigation that measured a 26 percent reduction in red oak seedling tree weight over a 16-week period (Dixon, 1988 as cited in Efroymson et al., 1997a). Moderate confidence in the plant benchmark is assumed because it is based on 17 values from experiments conducted with a wide range of different plant species (Efroymson et al., 1997a).

Efroymson et al. (1997b) presents an alternative lead soil benchmark of 500 mg/kg based on earthworm studies conducted by Bengtsson et al. (1986), who examined the effects of lead on *Dendrobaena rubida* at different acidities. After 4 months at pH 4.5, the number of cocoons produced per worm, hatchlings/cocoon, and percent hatched cocoons were reduced 75, 100, and 100 percent, respectively, by 500 ppm lead, while 100 ppm had no effect. Confidence in this benchmark is low due to the limited amount of data (Efroymson et al., 1997b).

Another lead soil benchmark of 900 mg/kg, presented by Efroymson et al. (1997b), is based on the 10th percentile of 36 reported effective values reported on the effects to soil microbial processes. Confidence in this benchmark is high because of the relatively large amount of data available for a variety of functional measures (Efroymson et al., 1997b).

Mercury. The ecological EBSV value of 0.1 mg/kg for mercury is based on survival of earthworms (*Otochaetus* sp.) in a soil/dung mixture after the application of a soluble form of mercury ($HgCl_2$) and assumes 100 percent bioavailability (Abbasi and Soni, 1983 as cited in

Efroymson et al., 1997b). The confidence in this EBSV is low due to the limited amount of data (Efroymson et al., 1997b).

Alternative soil benchmark values include 30 mg/kg for soil microbial processes (Efroymson et al., 1997b) and 0.3 mg/kg for phytotoxic effects (Efroymson et al., 1997a). The confidence in the microbial process is high because of the relatively large amount of data availability for a variety of functional measures (Efroymson et al., 1997b). However, the confidence in the pytotoxicity benchmark is low because it was based on a secondary reference, and the toxicity threshold for another study was more than two orders of magnitude higher (Efroymson et al., 1997a).

Zinc. The EPA Region 4 benchmark screening criteria of 50 mg/kg for zinc is based on effects to various plant species. As presented by Efroymson et al. (1997a), confidence in this benchmark is moderate. The benchmark represents the 10th percentile of multiple studies.

Efroymson et al. (1997b) present a 200 mg/kg benchmark based on several earthworm studies. The benchmark is somewhat lower than concentrations at which effects have been observed. The low concentrations include: measures of cocoon production of *E. fetida* in two artificial soils (136 ppm, 142 ppm) (Spurgeon and Hopkin, 1996) and 132 ppm, which is the LC_{50} for *E. fetida* divided by 5 (Neuhauser et al., 1985). Confidence in this benchmark is moderate (Efroymson et al., 1997b).

Efroymson et al. (1997b) also present a 100 mg/kg benchmark based on the 10th percentile of 46 reported soil microbial studies. Confidence in this benchmark is high because of the relatively large amount of data available for a variety of functional response factors (Efroymson et al., 1997b).

Cymene. There is no Region 4 EBSV for this VOC, nor could an alternate benchmark be located in available literature.

Benzoic Acid. There is no Region 4 EBSV for this SVOC, nor could an alternate benchmark be located in available literature.

Carbazole. There is no Region 4 EBSV for this SVOC, nor could an alternate benchmark be located in available literature.

PAHs. The EBSV of 0.1 mg/kg for benzo(a)pyrene, fluoranthene, phenanthrene, and pyrene represent Dutch soil detection limits and do not represent any type of measured biological response, but rather "were selected because they provide the only guidance available" (Beyer, 1990). No alternate benchmarks were located.

4,4'-DDT and Metabolites. The EBSVs for DDT and its metabolites, 0.0025 mg/kg, are presumably based on Dutch soil detection limits and may not represent any type of measured biological response (Dutch Ministry of Housing, Spatial Planning, and Environment [MHSPE], 1994), though this could not be confirmed. No alternate benchmarks were located.

alpha and gamma-Chlordane. The EBSVs of 0.1 mg/kg for chlordane isomers represent Dutch soil detection limits for aliphatic chlorinated hydrocarbons and may not represent any type of measured biological response, but rather "were selected because they provide the only guidance available" (Beyer, 1990). No alternate benchmarks were located.

Dieldrin. The EBSV of 0.0005 mg/kg for dieldrin (MHSPE, 1994) is presumably based on Dutch soil detection limits and may not represent any type of measured biological response, though this could not be confirmed. No alternate benchmarks were located.

Aroclor 1260. The Region 4 individual PCB EBSV of 0.02 mg/kg is based on total PCB, which is presumably based on Dutch soil detection limits and may not represent any type of measured biological response (MHSPE, 1994), though this could not be confirmed. Therefore, the uncertainty surrounding this criterion is significant, because soil detection limits in the Netherlands have no connection to biological effects either at the site or in the Netherlands.

An alternative screening benchmark, presented by Efroymson et al. (1997b), of 40 ppm is based on a study by Strek and Weber (1980). The study investigated the effects of Aroclor 1254 on fescue, sorghum, corn, soybeans, and beets grown in a sandy soil from seed for 16 days. Fresh foliage weight was reduced at 40 ppm, while 20 ppm had no effect. Confidence in the benchmark of 40 ppm for PCBs is low (Efroymson et al., 1997b), because it is based on fewer than 10 values.

6.4.4.2 Surface Water

The majority of EPA Region 4 EBSVs, are derived from the Screening Workshop prepared by the Region 4 Water Management Division. These values were obtained from water quality criteria documents and represent the chronic water quality criteria values for the protection of

aquatic life. If there was insufficient information available to derive a criterion, the lowest reported effect level was used with the application of a safety factor of ten to protect the most sensitive species. A safety factor of ten was also used to derive a chronic value if only acute information was available. The origin and basis of the EPA Region 4 surface water EBSVs, as well as alternate benchmarks, if available, are as follows.

Cymene. There is no EPA Region 4 EBSV for this VOC, nor could an alternate benchmark be located in available literature.

Bis(2-ethylhexyl)phthalate. The EBSV for this phthalate, <0.0003 mg/L, is based on the lowest reported effect level for two aquatic species with a safety factor of ten applied. Suter and Tsao (1996) present an alternative Tier II secondary chronic value of 0.003 mg/L.

6.4.4.3 Sediment

Region 4 sediment EBSVs are derived from statistical interpretation of effects databases obtained from the literature as reported in publications from the State of Florida, the National Oceanic and Atmospheric Administration, and a joint publication by Long et. al. These observations are generally based on observations of direct toxicity. When the Contract Laboratory Program's (CLP) practical quantification Limit (PQL) is above the effect level, the screening level value defaults to the PQL. Region 4 has also adopted other agency (e.g., EPA Region 5 and EPA Region 3) screening levels. The origin and basis of the EPA Region 4 sediment EBSVs, as well as alternate benchmarks, if available, are as follows.

Acetone. The Region 4 EBSV for acetone, 0.453 mg/kg, was adopted from the 1999 EPA Region 5 ecological data quality objectives (EDQL). It is unclear whether this value was derived from sediment quality criteria or if it was developed using the equilibrium partitioning (EQP) approach.

Cymene. There is no Region 4 EBSV for this compound, nor could an alternate benchmark be located in available literature.

Benzo(a)pyrene. The EBSV for this PAH, 0.33 mg/kg, is based on the CLP PQL because the effects value calculated by MacDonald (1994) was less than the PQL.

Benzoic acid. The Region 4 EBSV for this compound, 0.65 mg/kg, was adopted from the 1995 EPA Region 3 Biological/Technical Assistance Group screening levels. It is unclear how this value was derived.

4,4'-DDE. The 0.0033 mg/kg EBSV for 4,4'-dichlorodiphenyldichloroethane (DDD), 4,4'-DDE, and 4,4'-dichlorodiphenyltrichloroethane (DDT) is based on the CLP PQL because the effects value derived by Long and Morgan (1991) was less than the PQL.

PCBs. The 0.033 mg/kg EBSV for PCBs (0.067 mg/kg for Aroclor 1221) is based on the CLP PQL for total PCBs because the effects value derived by MacDonald (1994) was less than the PQL.

6.4.5 Chemical Characteristics

It is important to consider the characteristics of site contaminants in the assessment of risk. General categories of COPECs on site include inorganic analytes, VOCs, SVOCs, pesticides, and PCBs.

Inorganic Analytes. This class of COPEC does not tend to degrade in the environment, although they may change physical valance or species. They are generally considered to be indefinitely persistent. Dissolved inorganic metals can interact with soil or other solids by ion exchange, adsorption, precipitation, or complexation and can act as catalysts in biodegradation processes of other chemicals. These physiochemical processes are affected by:

- pH
- Composition of groundwater in the unsaturated zone
- Redox conditions
- Type and amount of organic matter, clay minerals, and oxyhydroxide minerals.

In general, southeastern soils contain abundant organic matter of significant importance to limit the availability of inorganic compounds. The pH of soil at LFNAS has not been determined, but acidic soils favor dissolution of carbonate and oxyhydroxide compounds, which can release a variety of inorganic compounds from their mineral structure or their surfaces.

Metallic ions may be bound to soil particulates by a combination of forces ranging from electrostatic to covalent forces (Mortland, 1985). When stronger covalent bonding dominates, certain ions are specifically bound and the reversibility of exchange decreases. This type of bonding may occur in organic matter, clays, and hydrous oxides (Roy, et al., 1989; Scrivner, et al., 1986; Gerritse and van Driel, 1984), all of which may be present in LFNAS soil in significant amounts (Roy, et al., 1989; Scrivner, et al., 1986; Gerritse and van Driel, 1984).

In soil, most heavy metals become less mobile with an increase in pH. This observation can be explained by a number of reactions:

- Precipitation of heavy metal hydroxides (Sposito, 1984)
- Changes in the carbonate and phosphate concentrations in the soil/groundwater (Huang et al., 1977)
- Adsorption and desorption of metals by hydrous oxides (Aiken et al., 1985)
- Formation of iron and manganese oxides (Suarez and Langmuir, 1976; Murray, 1975).

All natural soils contain trace levels of metals. Therefore, the presence of metals in soil may not be indicative of contamination. This trace level of metal concentration, known as background levels, is primarily related to the parent material(s) from which the soil was formed. The basic environmental concern of inorganic compounds in soil is when the metal is in its soluble form. Metals associated with the aqueous phase of soils are subject to migrate with groundwater and may be transported through the unsaturated zone to groundwater. Additionally, ionized metals are more bioavailable to ecological receptors.

Volatile Organic Compounds. VOCs released to surface soil are likely to be removed by volatilization. VOCs were also detected in surface water and sediment. Volatilization is the most important removal process for VOCs in surface water. Biodegradation may occur, but at a much slower rate.

Semivolatile Organic Compounds. Identified SVOCs include four PAHs (benzo[a]pyrene, fluoranthene, phenanthrene, and pyrene). Individual PAH compounds are distinguished from each other by the number and position of the aromatic rings and the position of substitution on the ring system. The higher molecular weight PAHs (e.g., benzo[a]pyrene) are known to be carcinogenic, mutagenic, and teratogenic to a wide variety of organisms (Eisler, 2000). However, they are typically immobile because of their large molecular volumes and their

extremely low volatability and solubility. Lower molecular weight PAHs (e.g., pyrene and fluoranthene) are generally not carcinogenic, but can be more acutely toxic than their higher molecular weight relatives (Eisler, 2000). Additionally, lower weight PAHs are absorbed by plants more readily than higher molecular weight PAHs (Eisler, 2000). PAHs show little tendency to biomagnify in food chains because most are rapidly metabolized and have a relatively short half-life in plants and animals (Eisler, 2000).

Organochlorine Pesticides. Organochlorinated pesticides, such as 4,4'-DDT and its metabolites, absorb very strongly to soil and sediment, but they may evaporate as well as photodegrade at the soil surface. Pesticides can accumulate and concentrate in fat tissue, but plants do not readily take up most pesticides. In birds and mammals, the most significant route of exposure is oral. Because invertebrates, such as earthworms, tend to accumulate pesticides, insectivorous mammals may be exposed to higher concentrations than mammals that feed heavily on plants, which tend not to accumulate pesticides in their tissues. Insectivorous mammals generally contain higher concentrations of pesticides than herbivorous mammals because many pesticides are accumulative.

Polychlorinated Biphenyls. PCBs sorb tightly to soil and sediment and are resistant to biodegradation. PCBs are known to accumulate in some organisms and are highly lipophilic. In birds and mammals, PCBs may disrupt normal patterns of growth, reproduction, and metabolism. Mammals have a greater ability to metabolize PCBs than birds, but birds are generally less susceptible to the acute effects of PCBs than mammals.

6.5 SLERA Discussion

Results of the initial screening exercise (Section 6.3.2) identified multiple COPECs in surface soil, surface water, and sediment. However, as presented in Section 6.4, there are several factors that must be considered prior to identifying these chemicals as potential risk drivers.

6.5.1 Landfill Area 1 Discussion

6.5.1.1 Surface Soil

Inorganic Analytes. Based on the analysis of background, the MDC of aluminum is less than the BSC. Therefore, this analyte is not recommended for further investigation.

Based on the geochemical evaluation, the level of arsenic, barium, cadmium, calcium, chromium, copper, iron, lead, manganese, mercury, nickel, potassium, sodium, vanadium, and zinc at the landfill are naturally occurring. Therefore, these analytes are not recommended for further investigation.

Based on the analysis of background, antimony was the only inorganic analyte that may contain a component of contamination. However, this metal is present at anomalously high concentrations at only one sample location (LF1SB06) out of 20, and the likelihood of exposure to excessive levels of antimony is minimal because the probability that a receptor will be exposed is low. Therefore, this metal is not recommended for further investigation.

VOCs. Cymene was the only VOC selected as a COPEC due to the lack of an EBSV. However, given that no other VOCs were detected at the site, and this compound is not expected to bioaccumulate, cymene is not recommended for further investigation.

SVOCs. Benzo(a)pyrene, fluoranthene, phenanthrene, and pyrene were each detected at concentrations that exceeded their respective EBSVs and BSCs at four sample locations (LF01SS05, LF1SB02, LF01SB04, and LF1SB06). Although this class of compound does not readily bioaccumulate and the EBSVs are based on Dutch soil detection limits, the clustering of the detections in the eastern half of the site may be indicative of localized contamination. It is recommended that further investigation of these PAHs be completed using less conservative assumptions and/or other risk analysis techniques in order to clarify the presence or absence of potential ecological risk.

Pesticides. DDT and its metabolites, DDD and DDE, were detected fairly uniformly across the site. Although the EBSVs for these pesticides are likely based on Dutch soil background concentrations, no alternate screening values were available. Therefore, given the uncertainty regarding potential risk to terrestrial receptors, it is recommended that additional investigation be completed using less conservative assumptions and/or other risk analysis techniques in order to clarify the presence or absence of potential ecological risk.

Alpha- and gamma-chlordane were detected at elevated concentrations at only one sample location (LF1SS05) out of 20. Although these pesticides are considered bioaccumulative, given the likelihood of exposure is minimal due to infrequent detection, and that the surface soil levels only slightly exceeded EBSVs (HQ_{screen} values = 3.6 and 3.7, respectively), these compounds are not recommended for further investigation.

Dieldrin exceeded the EBSV 3 out of 20 times in the eastern portion of the landfill (LF1SS01, LF1SS03, and LF1SS05). Although the incidence of exceedance is relatively low, the magnitude of exceedance is high (HQ_{screen} range 14.2 to 386), and although the basis of the EBSV is unknown, there are no available alternative values to assess screening risk. Therefore, it is recommended that dieldrin be investigated further using less conservative assumptions and/or other risk analysis techniques in order to clarify the presence or absence of potential ecological risk.

PCBs. Aroclor 1254 and 1260 exceeded EBSVs in 2 out of 20 and 13 out of 20 samples, respectively. The EBSV (0.02 mg/kg) used for both Aroclors is based on Dutch soil detection limits for total PCBs. An alternative EBSV of 40 mg/kg for Aroclor 1254 is available, which would result in HQ_{screen} values less than 1, but there is low confidence in the value and it may or may not be effective in evaluating Aroclor 1260. Therefore, because of the widespread distribution of PCBs as well as a high magnitude of exceedance, it is recommended that additional investigation of these two Aroclors be completed using less conservative assumptions and/or other risk analysis techniques in order to clarify the presence or absence of potential ecological risk.

6.5.1.2 Surface Water

Inorganic analytes were the only chemicals detected in surface water at Landfill Area 1. The only metals that exceeded their respective EBSVs and BSCs were chromium and thallium (LF1SW02) and lead (LF1SW04). However, based on geochemical evaluation, all three metals are shown to be naturally occurring. Therefore, it is recommended that chromium, lead and thallium be excluded from further investigation.

6.5.1.3 Sediment

Four metals (copper, lead, mercury, and zinc) exceeded their respective EBSVs and BSCs at LF1SD04. In the same sample, DDE, Aroclor 1254, and Aroclor 1260 exceeded their EBSVs. Additionally, two metals (calcium and vanadium) were identified as COPEC due to the lack of EBSVs and exceedances of BSCs.

Calcium, copper, lead, mercury, vanadium, and zinc were initially identified as COPEC; however, based on geochemical evaluation, all are shown to be naturally occurring. Therefore, these inorganic analytes are not recommended for further investigation.

Although 4,4'-DDE, Aroclor 1254, and Aroclor 1260 only slightly exceeded their EBSVs, HQ_{screen} 2.9, 3.8, and 4.6, respectively, these chlorinated compounds are considered to be highly bioaccumulative. Therefore, all three are recommended for further investigation using less conservative assumptions and/or other risk analysis techniques in order to clarify the presence or absence of potential ecological risk.

6.5.2 Landfill Area 3 Discussion

6.5.2.1 Surface Soil

Inorganic Analytes. Based on the analysis of background, the MDC of aluminum and vanadium are less than the BSC. Therefore, these analytes are not recommended for further investigation.

Based on the geochemical evaluation, the concentrations of calcium, chromium, iron, potassium, and sodium, at the landfill are naturally occurring. Therefore, these analytes are not recommended for further investigation.

Based on the analysis of background, copper and mercury may contain a component of contamination. However, these metals are present at anomalously high concentrations at only one sample location (LF3SS03 and LF3SS04, respectively) out of 20, and the likelihood of exposure to excessive levels of copper and mercury is minimal due to the low frequency of detection. Therefore, these metals are not recommended for further investigation.

Based on the analysis of background, lead may contain a component of contamination. However, alternative screening values of 500 mg/kg and 900 mg/kg, based on effects to earthworms and soil microbial processes, yield HQ_{screen} values of 0.4 and 0.2, respectively. Given that the alternative HQ_{screen} values are less than 1, the EBSV HQ_{screen} exceeds 1 only slightly (3.7), and only 2 of 20 locations exceeded the EBSV, lead is not recommended for further evaluation.

Based on the analysis of background, zinc may contain a component of contamination. However, alternative screening values of 200 mg/kg, based on effects to earthworms, and 100 mg/kg based on soil microbial processes, yield HQ_{screen} values of 0.8 and 1.9, respectively. Given that the soil microbial alternative HQ_{screen} value and the EBSV HQ_{screen} value (3.7) exceed

1 only slightly, and the earthworm alternative HQ_{screen} value is less than 1, zinc is not recommended for further evaluation.

VOCs. Cymene was the only VOC selected as a COPEC due to the lack of an EBSV. However, given that no other VOCs detected at the site had HQ_{screen} values greater than 1, and this compound is not expected to bioaccumulate, cymene is not recommended for further investigation.

SVOCs. Based on the analysis of background, the MDC of anthracene is less than the BSC. Therefore, this PAH is not recommended for further investigation.

Benzo(a)pyrene, fluoranthene, phenanthrene, and pyrene were each detected at concentrations that exceeded their respective EBSVs and/or BSCs at two sample locations (LF3SS02 and LF3SS11), and benzo(a)pyrene, fluoranthene, and pyrene exceeded their EBSVs and/or BSCs at LF3SB05. Given that this class of compound does not readily bioaccumulate, the EBSVs are based on Dutch soil detection limits, and the spatial distribution of exceedances are not indicative of hotspots or widespread contamination, further investigation of these PAHs is not recommended.

Benzoic acid and carbazole were selected as COPECs due to the lack of EBSVs. However, given that these compounds are not expected to bioaccumulate and spatial distribution is not indicative of a hot spot or widespread contamination, these compounds are not recommended for further investigation.

6.5.2.2 Surface Water

Inorganic analytes were the only chemicals detected in surface water at Landfill Area 3. The only metals that exceeded their respective EBSVs and BSCs were chromium and lead. However, based on geochemical evaluation, concentrations of these two metals were shown to be naturally occurring. Therefore, it is recommended that these analytes be excluded from further investigation.

6.5.2.3 Sediment

Inorganic Analytes. No EBSVs were available for aluminum, barium, beryllium, calcium, magnesium, manganese, potassium, selenium, sodium, and vanadium. However, none of these metals exceeded their respective BSCs. Additionally, based on geochemical evaluation, mercury

was shown to be naturally occurring. Therefore, further investigation of these analytes is not recommended.

Iron exceeded its BSC in at one sample location (LF3SD04) by almost three times the BSC. No EBSV is available for this analyte, nor could an alternate value be located. Due to the magnitude of background exceedance, it is recommended that this metal be examined further using less conservative assumptions and/or other risk analysis techniques in order to clarify the presence or absence of potential ecological risk.

VOCs. Acetone was detected in 3 out of 4 sample locations, but exceeded the EBSV at only one location. Given the small magnitude of exceedance (HQ_{screen} = 1.05) and that acetone is nonbioaccumulative, further examination of this VOC is not recommended.

Cymene was detected in one of four sample locations. This VOC was selected as a COPEC due to the lack of an EBSV. However, given that this compound is not expected to bioaccumulate, further investigation of this compound is not recommended.

SVOCs. Chrysene, fluoranthene, and pyrene exceeded their EBSVs in 1 of 4 locations. However, all detections of these PAHs were less than their BSCs. Therefore, these compounds are not recommended for further investigation.

Benzo(a)pyrene and benzoic acid both exceeded their EBSV in 1 out of 4 samples (LF3SD03 and LF3SD01, respectively). However, neither compound is considered to be bioaccumulative, and the magnitude of exceedance of these SVOCs is minimal (HQ_{screen} values = 1.35 and 1.19, respectively). Therefore, neither benzo(a)pyrene nor benzoic acid is recommended for further investigation using less conservative assumptions and/or other risk analysis techniques in order to clarify the presence or absence of potential ecological risk.

PCBs. Aroclor 1260 was detected at three sample locations, but exceeded the EBSV at only two locations (LF3SD02 and LF3SD03). Although the HQ_{screen} magnitude of exceedance is considered minimal, this compound is considered highly bioaccumulative and is recommended for further investigation using less conservative assumptions and/or other risk analysis techniques in order to clarify the presence or absence of potential ecological risk.

6.6 Risk Management Recommendations

Based on the SLERA, analytes and compounds at Landfill Areas 1 and 3 were identified as having the potential to pose unacceptable risk to the environment. Therefore, additional investigation of these chemicals is warranted in order to determine whether these contaminants should be classified as COC. To facilitate the evaluation, it is recommended that food chain modeling and less conservative exposure assumptions be conducted to further evaluate potential risk.

6.6.1 Landfill Area 1 Recommendations

As presented in Section 6.3.2.1, the Landfill Area 1 SLERA initially identified 30 surface soil, 17 surface water, and 16 sediment COPECs. Based on discussion presented in the problem formulation, the following chemicals may pose unacceptable risk:

Surface Soil	Sediment
Benzo(a)pyrene	4,4'-DDE
Fluoranthene	Aroclor 1254
Phenanthrene	Aroclor 1260
Pyrene	
4,4'-DDD	
4,4'-DDE	
4,4'-DDT	
Dieldrin	
Aroclor 1254	
Aroclor 1260	

Based on SLERA discussion, it is recommended that risk from chemicals detected in surface water be considered acceptable. Therefore, no further evaluation is warranted.

6.6.2 Landfill Area 3 Recommendations

As presented in Section 6.3.2.2, the Landfill Area 3 SLERA initially identified 19 surface soil, 11 surface water, and 20 sediment COPECs. Based on discussion presented in the problem formulation, the following chemicals in sediment may pose unacceptable risk:

Sediment
Iron
Aroclor 1260

Based on SLERA discussion, it is recommended that risk from chemicals detected in surface soil and surface water be considered acceptable. Therefore, no further evaluation is warranted.

7.0 Summary and Conclusions

7.1 Summary

7.1.1 Landfill Area 1

Shaw conducted a RI for the area known as Landfill Area 1 at the former LFNAS. The RI consisted of a historical records search, historical aerial photograph review, test pit excavation, piezometer and monitoring well installation, surface and subsurface soil sampling, groundwater sample collection, a comparison of Landfill Area 1 data to established background environmental media sediment and surface water concentrations, completion of a BHHRA, and completion of a SLERA.

Landfill Area 1 encompasses approximately 6.2 acres of land near the southeastern corner of the former LFNAS property. Today, the site is an inactive, heavily vegetated parcel bordered to the north by Wildwood Road and a drainage ditch paralleling the road to the north; south and east by a wetland area; and west by a low-flat wooded area that contains standing water during the rainy season. The surface of the landfill is mounded up to 10 feet above the surrounding terrain in places due to past burial of debris. Debris is widely visible at the surface throughout the landfill. Common types of debris seen at the surface include: scrap metal, glass, concrete, rusted drums, fiberglass materials, power poles, and other miscellaneous debris.

Historical LFNAS records indicate that the area known as Landfill Area 1 was referred to as a "Sanitary Yard Dump" or "Main Dump" on several maps dating from the 1940s and 1950s. A former employee at LFNAS reported that the site was formerly used to store compost and creosote treated poles during the years when the Navy operated the Base. No other details concerning historical use of the site were available. A review of historical aerial photographs first indicated activity at Landfill Area 1 between 1947 and 1948. The landfill began as a small circular feature in the northeast corner of the area now known as Landfill Area 1. The landfill remained similar in appearance until approximately 1954, when its size nearly doubled. The 1958 aerial photo showed the landfill area had been cleared to its maximum length. Based on the 1969 aerial, very little if any activity occurred between 1958 and 1969. Post-DOD activity at the landfill was documented by aerial photographs from approximately 1971 through 1990. During that time period the width of Landfill Area 1 was expanded into parts of the surrounding wetland area.

Surface soil samples were collected from twenty locations within and outside the presumed landfill boundary. Since the landfill is directly adjacent to Three Mile Swamp on the east and south sides, some of the soil samples planned for outside the landfill boundaries were collected from the landfill. The analytical results indicated the presence of five SVOCs, two PCB compounds, one pesticide, eight inorganic, and TRPH concentrations exceeding background concentrations and residential SCTLs at thirteen sample locations.

Subsurface soil samples were collected from locations coinciding with eight of the surface samples, outside or near the perimeter of the landfill boundary. The results indicate the presence of five SVOCs, one PCB compound, one pesticide, five inorganics, and TRPH concentrations exceeding background and residential SCTLs at locations LF1SB04, -SB05, and -SB06. These locations are widely separated, but all occur in the south-central or southeast portion of Landfill Area 1.

Sediment samples were collected from four locations, three in the drainage swale north of Wildwood Road, and one from the creek/wetland area east of Landfill Area 1. All four samples contained concentrations of various compounds exceeding the SQAGs. Some of the exceedances include: VOCs, SVOCs, pesticides, PCBs, and three inorganics (copper, lead, and mercury).

Surface water samples were collocated with the sediment samples and analyzed for the same parameters. Chromium was the only analyte detected in the surface water in excess of the SWCTL, at two locations.

During the RI, groundwater samples were collected from two depth intervals at six piezometer locations by direct-push. Following installation of a deep well, conversion of one piezometer to a permanent monitoring well, and installation of four shallow wells, groundwater samples were collected from all existing wells. The direct-push results indicated bis(2-ethylhexyl)phthalate and fluoranthene that exceeded GCTLs in four locations. Analysis of groundwater samples from the monitoring wells indicated the presence of vinyl chloride in deep well LF1DW01 and chlorobenzene in well LF1MW02 that exceeded GCTLs. A resample of the deep well confirmed the exceedance of the GCTL for vinyl chloride. Total metals exceeding GCTLs were detected in all Landfill Area 1 wells.

Exceedances for inorganics were compared to background concentrations established during the background chemical data study. The data sets were subjected to statistical tests and if necessary, geochemical evaluations were performed to determine if compounds detected were contaminant-related. A review of the soil data set (surface and subsurface) indicated anomalous concentrations of antimony that may contain a component of contamination. When subjected to geochemical evaluation, inorganics detected in the sediment and surface water samples were not determined to be contaminant-related. In groundwater, arsenic, iron, and manganese were determined to be elevated due to reductive dissolution, as a secondary effect of organic contamination. Zinc may contain a component of contamination.

The BHHRA identified antimony, PCBs, dieldrin, several PAHs, and TRPH as COPCs for surface soil at Landfill Area 1. The COPCs for total soil are similar to those for surface soil, except PCB and PAH concentrations are higher in the subsurface soil. COCs identified in soil include several PAHs, Aroclor 1254, Aroclor 1260, and dieldrin. No COPCs were selected for the sediment. For surface water, only TRPH was selected as a COPC. The groundwater evaluation identified the following COPCs: iron, manganese, bis(2-ethylhexyl)phthalate, TRPH, chlorobenzene, and vinyl chloride. COCs for groundwater include bis(2-ethylhexyl)phthalate, chlorobenzene, vinyl chloride, TRPH, and manganese.

The SLERA identified twenty-six COPEC (fourteen inorganics, four PAHs, six pesticides, and two PCBs) for surface soil; ten inorganics analytes were selected as COPEC for surface water; and seven COPEC (four inorganic compounds, one pesticide, and two PCBs) for sediment. Based on further evaluation the following compounds and associated media warrant further investigation/evaluation to determine their potential ecological risk: surface soil (benzo(a)pyrene, fluoranthene, phenanthrene, pyrene, 4,4'-DDD, 4,4'-DDE, 4,4'DDT, dieldrin, Aroclor 1254, and Aroclor 1260) and sediment (4,4-DDE, Aroclor 1254, and Aroclor 1260). Metals identified as COPEC were determined to be naturally occurring and were excluded as a concern.

7.1.2 Landfill Area 3

Shaw conducted a RI for the area known as Landfill Area 3 at the former LFNAS. The RI consisted of a historical records search, historical aerial photograph review, test pit excavation, piezometer and monitoring well installation, surface and subsurface soil sampling, groundwater sample collection, a comparison of Landfill Area 3 data to established background environmental media concentrations, completion of a BHHRA, and completion of a SLERA.

Landfill Area 3 encompasses approximately 7.4 acres of land near the southern edge of the former LFNAS property. Today, the site is an inactive, wooded parcel bordered on the north, west, and east by a thick forest; and to the south by a dirt access road and a wooded area. In 2001, a portion of the landfill was clear cut to remove large timber. These areas have since become overgrown with scrub vegetation, where sunlight is more prevalent. The surface of the landfill is relatively flat with three areas of mounded soil. Scattered debris is visible at the surface from the south-central portion of the landfill to the east and northeast. Common types of debris seen at the surface include: scrap metal, glass, a 1950s vintage automobile body, and other miscellaneous debris.

Historical LFNAS records indicate that the area known as Landfill Area 3 was referred to as a "Sanitary Land Fill" on a map dated 1957. The area was reportedly used to dispose of domestic waste, refuse, and medical waste. No other details concerning historical use of the site were available. A review of historical aerial photographs first indicated landfill-related activity at Landfill Area 3 in 1951. Prior to that time, aerial photographs indicate activity adjacent to the future landfill as early as 1944. Some type of industrial plant or operation was present just south of the landfill boundary from approximately 1944 to 1948, but the function of the facility is unknown. Activity at the landfill continued through at least 1958. The next available aerial photo (1969) indicates some disturbance and what appears to be additional debris. Additional disposal may have occurred at the site after the Navy relinquished control of the Base in 1963, but this could not be confirmed. No other activity was observed at Landfill Area 3 after approximately 1969.

Twenty-five test pits were excavated in an attempt to define the boundaries of the landfill. Several of the test pits, including 9, 14, 15, and 23, encountered debris at depth ranging from 3 to 7 feet bgs. Subsurface debris encountered includes: gas masks, cartridges, glass, trash cans, mufflers, and other miscellaneous debris.

Surface soil samples were collected from twenty locations within and outside the presumed landfill boundary. The analytical results indicated the presence of one VOC, two SVOCs, and one inorganic (copper) compound exceeding background concentrations and residential SCTLs at nine sample locations.

Subsurface soil samples were collected from locations coinciding with eight of the surface samples, outside or near the perimeter of the landfill boundary. The results indicate the presence of detectable VOCs, SVOCs, inorganics, and TRPH concentrations, but none in excess of background and/or residential SCTLs.

Sediment samples were collected from four locations within Landfill Area 3. Sample LF3SD03 contained five PAHs, PCBs (Aroclor 1260), and mercury concentrations that exceed the SQAGs. The other three samples also exceed the SQAGs for PCBs.

Surface water samples were collocated with the sediment samples and analyzed for the same parameters. The SVOC bis(2-ethylhexyl)phthalate is present in three of the surface water samples at concentrations exceeding the SWCTL. Seven inorganics were detected above the SWCTLs in LF3SW03.

During the RI, groundwater samples were collected from two depth intervals at six piezometer locations by direct-push. The piezometers were placed on the perimeter of the landfill to determine the groundwater flow direction and collect groundwater screening samples. Concentrations of chlorinated hydrocarbons in excess of GCTLs were detected at depth in one direct-push sample (LF3PZ02 14 to 18 feet). During the Phase II investigation, additional groundwater samples were collected by direct-push to define the extent of chlorinated hydrocarbons detected previously. The Phase II sampling indicated the contaminant plume is localized.

Two of the piezometers were converted to permanent monitoring wells. In addition, four shallow wells and two deep wells were installed. Groundwater samples were collected from all existing wells. Analysis of groundwater samples from the monitoring wells indicated the presence of chlorinated hydrocarbons in excess of the GCTLs in well LF3MW06, which replaced destroyed piezometer LF3PZ02. Five inorganic compounds were detected that exceeded GCTLs.

Exceedances for inorganics were compared to background concentrations established during the background chemical data study. The data sets were subjected to statistical tests and if necessary, geochemical evaluations were performed to determine if compounds detected were contaminant-related. A review of the soil data set (surface and subsurface) indicated anomalous concentrations of cadmium, copper, lead, mercury, nickel, silver, and zinc that may contain a component of contamination. Both arsenic and iron were detected in sediment at anomalously high concentrations and possibly contain a component of contamination. Inorganics detected in the surface water samples were determined to be noncontaminant-related. In the groundwater, barium, chromium, iron, nickel, and manganese concentrations are anomalously high and may be contaminant-related. Iron may be elevated due to reductive dissolution, as a secondary effect of organic contamination.

The BHHRA identified copper, mercury, and two PAHs as COPCs for surface soil at Landfill Area 3. The COPCs for total soil are identical to those for surface soil, with higher concentrations in the surface soil. COCs identified in soil include benzo(a)anthracene and dibenz(a,h)anthracene. COPCs identified for the sediment include arsenic, iron, and benzo(a)pyrene. For surface water, only TRPH was selected as a COPC. The groundwater evaluation identified the following COPCs: iron, manganese, several VOCs and TRPH. COCs for groundwater include manganese, acetone, 1,2,4-trimethylbenzene, cis- and trans-1,2-DCE, TCE, vinyl chloride, and TRPH.

The SLERA identified nineteen COPEC (eleven inorganics, five PAHs, two SVOCs, and one VOC) for surface soil; thirteen COPEC (eleven inorganics, one VOC, and one SVOC) for surface water; and nineteen COPEC (twelve inorganic compound, two VOCs, four PAHs, one SVOC, and one PCB) for sediment. Based on further evaluation the following compounds and associated media warrant further investigation/evaluation to determine their potential ecological risk: sediment (iron and Aroclor 1260). None of the compounds identified as COPEC for the soil and surface water were deemed a potential ecological risk.

7.2 Conclusions

7.2.1 Landfill Area 1

The RI determined that abundant debris has been disposed of within the presumed boundary of Landfill Area 1. Based on personal accounts of past activities and a review of historical aerial photos, the Navy mainly utilized the eastern half of the landfill area for disposal of waste during the operational period of LFNAS. The nature of the waste disposal practices has been reported

by a previous employee; however, the types of debris were not confirmed. Due to the relatively unknown nature of the buried waste, an effort was made to avoid penetrating the landfill with equipment as much as possible. Test pits excavated on the western edge of the landfill did not encounter debris.

Post-DOD use of the landfill for disposal of demolition debris such as fiberglass and concrete has been documented during the 1970s and 1980s. Other types of debris such as drums have also been discovered at Landfill Area 1.

The soil and groundwater contamination detected is likely attributable to past disposal activities at the site. Use of the site from approximately 1948 to 1990 makes it difficult to distinguish between DOD and post-DOD contamination. The exceedances in the soil are widely scattered across the site; however, all occur in the eastern half of the landfill area (east of LF1SB06), which may be attributable to DOD-related activities. Contaminants detected including PCBs, pesticides, and PAHs are not uncommon in waste disposal areas, but the date of their release is difficult to determine. Arsenic concentrations exceeding the SCTL are widespread in the soil, but these detections have been determined to be naturally occurring. The groundwater is relatively free of contamination, with only minor exceedances of the GCTL for chlorobenzene and vinyl chloride. The vinyl chloride detection appears confined to the southeast corner of the landfill where it meets Three Mile Swamp. The vertical extent of the vinyl chloride plume has not been confined. The presence of vinyl chloride, a breakdown product of TCE, with no parent compounds detected indicates the source is not recent.

The BHHRA established soil and groundwater as the only media of concern at Landfill Area 1, due largely to the presence of PCBs, PAHs, and dieldrin in soil and chlorobenzene and vinyl chloride in groundwater. The SLERA identified ten compounds in surface soil and three in sediment which may pose an unacceptable risk.

7.2.2 Landfill Area 3

The RI determined that abundant debris has been disposed of within the presumed boundary of Landfill Area 3. Based on past records and a review of historical aerial photos, the Navy utilized Landfill Area 3 from approximately 1951 through closure of the Base in 1963. A facility that may have been industrial was observed just southeast of the future landfill area from approximately 1944 to 1948. The exact type of facility or its practices are unknown. The landfill was reportedly only used by the Navy during the operational period of LFNAS. Aerial photographs indicate some disposal activity occurred on the northeast portion of Landfill Area 3

between 1958 and 1969, but it is unknown whether this activity occurred prior to closure of LFNAS or post-DOD. The nature of the waste disposal types was reported to include refuse, domestic waste, and medical waste. Due to the relatively unknown nature of the buried waste, an effort was made to avoid penetrating the landfill with equipment as much as possible. Test pits were used to define the approximate landfill boundaries, which were found to correlate with the disturbed areas seen on aerial photographs. Several of the test pits encountered debris, all below the static water table. The type of debris discovered is consistent with a domestic use or sanitary landfill. No medical waste was apparent based on the limited excavations within the burial areas of the landfill.

Only three scattered exceedances of the SCTLs for benzo(a)pyrene and dibenz(a,h)anthracene were detected. Concentrations of PAHs and PCBs were detected in sediment collected from ponds within the landfill, which exceed the SQAGs. These compounds are commonly found in the vicinity of disposal areas.

The most significant levels of contamination were detected in a direct-push groundwater sample collected adjacent to a piezometer previously located just outside the southeast boundary of the landfill. This area coincides with the location of the facility observed on the aerial photos from 1944 to 1948. Historical aerial photos, records, and maps reviewed provided no information on the purpose of the facility or what chemicals may have been stored there. Concentrations of chlorinated hydrocarbons including TCE, cis- and trans-1,2-DCE, and vinyl chloride were detected at concentrations well above the GCTLs. The piezometer was replaced with a permanent monitoring well after the piezometer was destroyed. Analytical results from the well were generally consistent with those from the direct-push sample. A subsequent expanded groundwater investigation in the vicinity of the chlorinated hydrocarbon detection indicated a very localized plume, both horizontally and vertically. The presence of high levels of TCE and its breakdown products along with the apparent limited extent of the contaminant plume likely indicates the release is recent. Two possible release scenarios could have caused the contamination observed. First, the release could have occurred post-DOD, unrelated to the historical building. The second possibility is a delayed release could have originated from a buried source (such as a drum) related to DOD-era activities. An older release would be expected to show more breakdown of TCE to its daughter compounds such as vinyl chloride. An older release of chlorinated hydrocarbons would also have migrated both horizontally and vertically farther away from the source than is seen.

The BHHRA established soil and groundwater as the only media of concern at Landfill Area 3, due largely to the presence of PAHs in soil and VOCs and manganese in groundwater. The SLERA identified two compounds in sediment, which may pose an unacceptable risk.

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