US Army Corps
of Engineers

February 2006

Supplemental Remedial Investigation of the Upland Area
Final Report

Fort Totten Coast Guard Station
Formerly Used Defense Site
FUDS Property No. C02NY0057

Queens, New York

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EXECUTIVE SUMMARY

The United States Army Corps of Engineers (USACE) conducted a Supplemental Remedial Investigation (SRI) at the Fort Totten Coast Guard Station in Queens, New York during July and August 2004. This investigation represents the final phase of the remedial investigation project conducted under the USACE's Formerly Used Defense Site (FUDS) program, and was initiated in response to comments from the New York State Departments of Environmental Conservation (NYSDEC) and Health (NYSDOH). This report focuses entirely on the portions of the upland area of the FUDS and Building 615. It summarizes and evaluates the results of the investigation conducted in summer 2004. It also presents an updated quantitative human health and ecological risk assessment using both previous RI data and current SRI data to examine future use of the FUDS property.

The SRI is separate from previous work involving water, sediment and aquatic life in Little Bay. USACE issued a No Further Action Record of Decision (ROD) in 2003 calling for follow-up sampling three years after the ROD to confirm the previous results. Detailed information on Little Bay is presented in the *Record of Decision for Little Bay, FUDS Fort Totten Coast Guard Station, Queens, New York* (USACE 2003).

The SRI soil sampling program was conducted to obtain additional data on semivolatile organic compounds (SVOCs) and metals from the upland areas. Shallow and deep soil samples were collected from 11 new soil boring locations. The soil sample locations were selected based on the presence of higher concentrations of SVOC levels during the previous sampling and as requested by the NYDEC. In order to address previous detections of SVOCs in the groundwater near MW-4, a replacement monitoring well was installed and sampled. The final element of the SRI focused on ambient indoor air monitoring for mercury inside Building 615. Samples were taken with a real-time monitor and fixed-based samplers at heights of three feet and six feet above each of the two floors. During collection of indoor air samples, a floor drain inside of Building 615 was located. Sediment/sludge samples were collected from within the pipe. In addition, real time and fixed based indoor air sample were also collected at this location.

The soil sampling showed that some metals and SVOCs are present in the soil in the upland portion of the FUDS at concentrations that exceed state screening criteria. However, the human health risk assessment indicated that soil in the majority of the upland portion the FUDS does not present an unacceptable risk under the multiple scenarios evaluated and may be considered for unrestricted use. Some soil in the Fill Area of the FUDS presents an unacceptable hazard under a residential reuse scenario due to lead. Therefore, it may be inappropriate for future unrestricted reuse of the Fill Area unless actions are taken to limit exposure. However, reuse of the Fill Area for nonresidential uses may be appropriate.

It is the conclusion of the screening level ecological risk assessment that refinement of the hazards identified are not warranted and that there are no unacceptable threats to ecological receptors from exposure to chemicals in the soil in the upland portions of the facility.

The sample results indicate that the groundwater at the FUDS may not be appropriate for use as a potable water source for residential consumption.

The sample results for the indoor air sampling in Building 615 indicated that there were no detectable concentrations of mercury greater than the state screening level. No further investigation of the indoor air is warranted.

Finally, the additional drain pipe located in Building 615 does not discharge into Little Bay and does not contain mercury at concentrations that would volatilize into the building and pose a health concern to workers. Future work will address concerns regarding the drywell/vault at the end of the Building 615 drain pipe.

USACE will initiate a Focused Feasibility Study of the Fill Area soil.

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1.0 INTRODUCTION

This Supplemental Remedial Investigation (SRI) Report presents and discusses the results of the SRI conducted at the Fort Totten Coast Guard Station in Queens, New York during July and August 2004. The U.S. Army Corps of Engineers (USACE) performed the SRI for soil and groundwater within the upland area and indoor air for mercury in Building 615. This investigation represents the final phase of the remedial investigation project conducted under the Corps' Formerly Used Defense Site (FUDS) program, and was initiated in response to comments from the New York State Departments of Environmental Conservation (NYSDEC) and Health (NYSDOH). The state agencies' comments, submitted in response to a 2002 draft remedial investigation report, focused on the need to collect additional samples of soil, air and groundwater at certain locations on the FUDS property.

1.1 Purpose of Report

The supplemental investigation of the upland area is separate from previous work involving water, sediment and aquatic life in Little Bay. These investigations found no significant concentrations of mercury in Little Bay's ecosystem, and the USACE issued a Record of Decision (ROD) in 2003 calling for follow-up sampling three years after the ROD to confirm the previous results. Detailed information on Little Bay is presented in the *Record of Decision for Little Bay, FUDS Fort Totten Coast Guard Station, Queens, New York* (USACE 2003) and the draft *Remedial Investigation, FUDS Totten Coast Guard Station, Queens, New York* (USACE 2002), which will be finalized in fall of 2004.

This report focuses entirely on the Fill Area and Other Area as described in the draft RI report (USACE 2002), and Building 615. It summarizes and evaluates the results of the investigation conducted in summer 2004. It also presents a quantitative human health and ecological risk assessment using both previous RI data and current SRI data to examine future use of the FUDS property.

1.2 Background

A brief description and background are presented here. A more detailed description of the site and site history are found in the draft RI report (USACE 2002).

1.2.1 Site Description

The Fort Totten FUDS is located on the current US Coast Guard Station in northeast Queens County, Long Island, New York. The facility is situated on a peninsula extending out into Little Neck Bay and the East River portion of Long Island Sound. A general site location is shown in Figure 1-1, and a map of the Fort Totten Installation (US Army Reserve and Coast Guard portions) is presented in Figure 1-2.

1.2.2 Site History

The Department of Defense (DoD) acquired Fort Totten, a 146.75-acre property, between 1857 and 1943, for the coastal defense of Long Island Sound and the eastern entrance to the East River. Fort Totten also served as a post-Civil War hospital, an engineering school, and a training site for West Point Cadets. It is currently the Headquarters for the 77th Army Reserve Command. The Department of the Army conveyed 9.60 acres of the property to the U.S. Coast Guard, while still retaining ownership of the remaining 137.15 acres. This FUDS project is limited to the excessed portion (9.60 acres) of Fort Totten presently owned by the U.S. Coast Guard.

1.2.3 Previous Investigations in the Upland Area

The upland portion of the Ft. Totten FUDS has been the subject of several previous investigations, dating back to 1986 (USACE 1988; USACE 2002). Shallow samples were collected from the soil and sediment on the property and five groundwater monitoring wells were installed. USACE initiated a comprehensive Remedial Investigation (RI) in 1997 to determine the nature and extent of the contamination reported in earlier studies. The RI was conducted in two phases. Phase I was conducted from July of 1997 through August of 1998 by the USACE, and included the collection and analysis of soil, sediment, and groundwater samples. Wipe samples were collected from the walls of Buildings 619 and 624 and analyzed for pesticides. Wipe samples were also collected from Building 615 and analyzed for mercury. A removal action was completed by USACE inside Building 615 to remove a source of mercury consisting of sumps, drains, and discharge pipes (USACE 2002).

The USACE performed Phase II from November 1999 through August 2000 to obtain more detailed information about the site. The investigation involved the collection of sediment, groundwater, surface water, and surface soil samples from the same or similar locations as Phase I. In the spring of 2002, the USACE collected an additional round of groundwater samples from the five existing monitoring wells.

During Phases I and II, USACE had collected and analyzed 92 soil samples from 70 different locations in upland areas of the Coast Guard Station. Although no pesticides or PCBs were detected, the analysis showed that concentrations of PAHs and metals in some surface soil samples were greater than the screening concentrations used by the NYSDEC. Levels of metals detected in the groundwater were found to be in the range of those occurring naturally in the environment, however low levels of PAHs were found in the groundwater at one monitoring well.

1.2.4 Project Objectives for the Supplemental Investigation

The USACE conducted a supplemental investigation in summer 2004 to address data gaps and questions raised by the NYSDEC and NYSDOH regarding the upland areas and Building 615. The upland areas are defined as those areas on the FUDS property that are east of the Little Bay shoreline and west of the North Loop road. Figure 1-3 shows the locations of Building 615, the Fill Area and Other Areas that make up the upland portion of the FUDS property that are addressed in this SRI. Descriptions of these locations are provided below.

The Fill Area was created when the Army placed excavated soil in a low spot near the recreation field to eliminate periods of standing water. The area is approximately 110 ft by 250 ft. The soil came from near Buildings 118, 119, and 121, former and existing vehicle maintenance shops, on the Army-owned portion of Fort Totten (USACE 2002). Parts of those buildings' parking lots were being excavated. There is a baseball field next to the Fill Area and several open areas west of the ball field and Bayside Street. The Other Area is defined as the portions of the upland area other than the Fill Area and buildings (e.g., Building 209) specifically investigated in the initial RI.

Building 615 is located along the seawall adjacent to Little Bay. It was originally used as a torpedo and mine repair facility. The armaments contained mercury in their guidance systems, and when repair required removal of the mercury, it was disposed of through the sinks and floor drains. Mercury contaminated sumps, drains, and discharge pipes had been removed from the building.

There were three principal questions for the supplemental investigation that were driven by state comments on the Draft 2002 RI report. These questions were:

1. What is the nature and extent of the SVOC and metals contamination in the upper soil horizon of the Other Area and Fill Area?
2. What is the concentration of SVOCs in the groundwater near MW-4?
3. Is there mercury in the indoor air at a concentration greater than 0.001 mg/m^3 in Building 615?

To address the first question, a sampling program was developed to obtain additional data on levels of polycyclic aromatic hydrocarbons (PAHs) and certain metals from the upland areas. Soil samples were collected from 11 new soil borings. USACE obtained 22 additional soil samples from 11 locations and analyzed the samples for semivolatile organic compounds (SVOCs), including PAHs, and metals. The sampling locations were selected based on the presence of higher concentrations of SVOC during the previous sampling round (USACE 2004).

During previous RI investigations, one of the monitoring wells located near Building 615, MW-4, contained levels of SVOCs slightly above the State's groundwater screening criteria. Surface water runoff, which could be the source of the SVOCs, may have had an impact on this monitoring well. In order to address the second question regarding SVOCs in the groundwater near MW-4, a replacement monitoring well was installed during the supplemental RI. The new monitoring well was developed and a groundwater sample was taken and analyzed for SVOCs.

The final element of the investigation focused on ambient indoor air monitoring for mercury inside Building 615 in order to address the third question above. Samples were taken with a real-time monitor and fixed-based samplers at heights of three feet and six feet above each of the two floors.

The following project objectives were identified for the SRI:

- Collect sufficient soil, groundwater and indoor air data to supplement the existing risk assessment of the upland area.
- Determine if mercury is present in the breathing zone for workers and visitors in Building 615.
- Evaluate fate and transport pathways.
- Assess the hazards/risks posed by any contamination present and if necessary provide recommendations for further remedial efforts.

1.2.5 Potential Applicable or Relevant and Appropriate Requirements

The National Contingency Plan (NCP) (see 40 CFR 300.430[e]) specifies that remedial actions must attain federal standards, requirements, criteria, limitations, or more stringent state standards determined to be legally applicable or relevant and appropriate requirements (ARARs) to the circumstances at a given site. To be applicable, a state or federal requirement must directly and fully address the hazardous substance, the action being taken, or other circumstance at a site.

ARARs are used in conjunction with risk-based goals to govern response activities and to establish cleanup goals. When ARARs are absent or are not sufficiently protective, USEPA uses data collected from the baseline risk assessment to determine cleanup levels.

Whether or not a requirement is appropriate (in addition, to being relevant) will vary depending on factors such as the duration of the response action, the form or concentration of the chemicals present, the nature of the release, the availability of other standards that more directly match the circumstances at the site, and other factors [Section 300.400(g)(2)]. In some cases only a portion of the requirement may be relevant and appropriate.

The identification of relevant and appropriate requirements is a two step process; only those requirements that are considered both relevant and appropriate must be addressed at FUDS. Environmental laws and regulations generally fit into three categories: (1) those that pertain to the management of certain chemicals; (2) those that restrict activities at a given location; and (3) those that control specific actions. There are, therefore, three primary types of ARARs.

- **Chemical specific**-ARARs are usually health- or risk-based restrictions on the amount or concentration of a chemical that may be found in or discharged to the environment. Location-specific ARARs prevent damage to unique or sensitive areas, such as floodplains, historic places, wetlands, and fragile ecosystems, and restrict other activities that are potentially harmful because of where they take place.

- **Action-specific** - ARARs control remedial activities involving the design or use of certain equipment, or regulate discrete actions. Since remedial or removal actions are not yet being evaluated at this stage of the investigation, the primary focus of this task was to identify chemical specific ARARs.

- **Location-specific** - ARARs. Action-specific requirements will be identified at a later date.

The types of legal requirements applying to responses will differ to some extent depending upon whether the activity in question takes place onsite or offsite (the term "onsite" includes not only the contaminated area at the site, but also all areas in close proximity to the contamination necessary for implementation of the response action) .

The potential ARARs for the response activities at the upland area and Bldg 615 have been identified in Table 1.1.

1.2.6 To-Be-Considered Guidelines and Other Controls

A requirement that is not applicable may be relevant and appropriate if it addresses problems or pertains to circumstances similar to those encountered at a site. While legally applicable requirements must be attained, compliance with relevant and appropriate requirements is based on the discretion of the lead agency. The scope and extent of ARARs that may apply to a response action will vary depending on where remedial activities take place. Laws and requirements enforced by agencies other than the USEPA may also be applicable or relevant and appropriate at a site. In other cases, the response may incorporate environmental policies or proposals that are not applicable or relevant and appropriate, but do address site- specific concerns and are called “To-Be-Considered” (TBC) items.

Conditions vary widely from site to site; thus, ARARs alone may not adequately protect human health and the environment. When ARARs are not fully protective, the lead agent may implement other federal or state policies, guidelines, or proposed rules capable of reducing the risks posed by a site. Such policies, guidelines, or proposed rules, (while not legally binding since they have not been promulgated), may be used in conjunction with ARARs to achieve an acceptable levels of risk. TBC items are evaluated along with ARARs as part of the RI/FS conducted for each site to set protective cleanup levels and goals.

The following potential TBC guidelines have been identified for response activities at the upland area and Building 615:

- New York State Department of Environmental Conservation. 1998. *Ambient Water Quality Standards and Guidance Values and Groundwater Effluent Limitations*. Division of Water, Technical and Operational Guidance Series (TOGS) No. 111. June.

- New York State Department of Environmental Conservation. 1994. *Determination of Soil Cleanup Objectives and Cleanup Levels*, Technical Assistance Guidance Memorandum (TAGM) HWR-94-4046 (revised).

2.0 FIELD INVESTIGATIONS

Surface soil samples were collected during the 2004 Supplemental RI field work. A monitoring well (MW), MW-4R, was installed and a groundwater sample collected. Air monitoring for mercury was performed, both real-time with an on-site portable analyzer and with samples collected and analyzed by a laboratory. In addition, two sediment samples were collected from a drainpipe associated with Building 615.

The objective of the field program was to collect 11 soil samples at 2 – 12 inches bgs and 12 – 24 inches bgs; collect air samples, analyzing for mercury, from Building 615; and collect groundwater samples from MW-4R.

2.1 Soil Samples

On June 21, 2004, 22 soil samples were collected from the 11 locations specified in the NYSDEC's letter of February 6, 2004 (see Appendix J). The samples were collected with three-inch inner diameter split-spoons, using a mobile truck-mounted drill rig equipped with 4.25-inch inner diameter hollow stem augers.

Each of the 11 locations was sampled at 2 – 12 inches below ground surface (bgs) and 12 – 24 inches bgs. The 22 soil samples were analyzed for Target Compound List (TCL) SVOCs by USEPA SW-846 method 8270C and for Target Analyte List Metals (TAL) metals by USEPA SW-846 methods in the 6000 and 7000 series. The soil locations for the 2004 sampling event were selected based the SVOC concentrations found in the RI Phase 2 sampling. The sample locations are shown in Figure 2-1.

The initial laboratory results of sample Ad2004-SS-11-SH had a wide range of SVOC results between primary, duplicate and quality assurance (QA) sample. Therefore, that location (boring B-11) was resampled on 25 August 2004 at both the 2 – 12 inches bgs and 12 – 24 inches bgs depths. The samples were analyzed for SVOCs.

2.1.1 Soil Samples B-1 through B-9

Nine soil borings were collected in the upland portion of the Ft. Totten FUDS known as the Other Area. The samples are numbered Ad2004-SS-1-SH through Ad2004-SS-9-SH, for the shallow samples (2 – 12 inches bgs) and Ad2004-SS-1-DP through Ad2004-SS-9-DP for the deep samples (12 – 24 inches bgs). The number of the soil sample corresponds to their boring numbers (B-1 through B-9) in the Figure 2-1.

2.1.2 Fill Area Soil Samples 10 and 11

Two soil borings were excavated in the Fill Area and soil samples collected, 2 – 12 inches bgs and 12 – 24 inches bgs. The two soil samples were numbered Ad2004-SS-10-SH and Ad2004-SS-11-SH for the shallow samples and, Ad2004-SS-10-DP and Ad2004-SS-11-DP, for the deep samples. The number of the soil sample corresponds to the boring number (B-10 and B-11) in the Figure 2-1.

2.2 Monitoring Well 4R

Monitoring well installation was the first field activity. Utility clearances were performed by the field geologist prior to intrusive work being performed at the site. The new monitoring well was installed on July 23, 2004, in the vicinity of the existing monitoring well MW-4, shown in Figure 2-1. The existing well previously exhibited elevated concentrations of SVOCs; however, since the well was flush-mounted in the parking lot and has sustained damage to its casing and seal, it was abandoned in accordance with the Work Plan (USACE 2004). Groundwater is located at a depth of approximately six feet bgs at this location and is down gradient from the upland portion of the site.

The new shallow monitoring well, designated MW-4R, is screened to straddle the water table with 2.5 feet of screen extending above the water table, and 2.5 feet below, for a total of five feet of screen. The well has a nominal diameter of 4 inches and is constructed of polyvinyl chloride (PVC) with a continuously wrapped screen. A solid casing extends from the screen to just below the ground surface. The top of the casing is equipped with an easily removable cap. After the new monitoring well (MW-4R) was installed, it was developed on August 1, 2004. A full description of the monitoring well installation and development process is provided in Section 4.5 of the 2004 Work Plan (USACE 2004).

2.2.1 Groundwater Sample from MW-4R

The new well was sampled in accordance with the monitoring well sampling protocol provided in Section 4.6 of the Work Plan (USACE 2004). It was purged and sampled on August 4, 2004, using low-flow (minimal drawdown) groundwater sampling procedures. At the time of opening the flush mount seal (at road level), water was observed above the inner well cap. This water is assumed to be parking lot run-off. The water inside the well mounting was removed, sampled (Sample No. MW-4R-01-02), and not allowed to enter the well.

Purging of the well proceeded until stabilization of water quality indicator parameters (pH, temperature, specific conductance, dissolved oxygen, oxidation-reduction potential, and turbidity). (Stabilization is established by three consecutive readings in which: pH variation is less than 0.2 units, temperature variation is less than 0.5° C, and all other parameters are less than 10 percent variation.)

The collected groundwater sample (and duplicate and QA sample), was analyzed for TCL SVOCs by EPA SW-846 Method 8270C in GC Ion-Trap Mode.

2.3 Air Monitoring

At the request of the NYDEC, sampling for mercury in indoor air was conducted in Building 615, the location of a previous removal action (USACE 1998) to address mercury contamination in floor drains. Sampling was conducted with a real time portable analyzer and with fix-based sampling apparatus followed by off-site sample analysis.

2.3.1 Real-Time Mercury-in-Air Samples

On June 26, 2004, real-time mercury air samples were obtained from Building 615, using a Ohio Lumex's Zeeman Portable Mercury Vapor Analyzer, model RA-915+. Two pairs of background samples were taken at 3 feet and 6 feet above the ground, outside, near the front door of Building 615. Forty-one sample pairs were taken at 3 feet and 6 feet above the floor, inside Building 615. Four air samples were taken above a floor drain near the photography laboratory at 1-inch, 1 foot, 3 feet, and 6 feet above the floor. The sampling locations for the real-time air monitoring of mercury are shown in Figure 2-2a and 2-2b.

2.3.2 Laboratory Analyzed Mercury-in-Air Samples

Subsequently, the fixed, laboratory analyzed, mercury air sample collection stations were set up at some of the real-time air monitoring locations. There were ten such sampling locations inside Building 615, each with sampling points 3 feet and 6 feet above the floor. In addition, a pair of background samples was taken at 3 feet and 6 feet above the ground, outside, near the front door of Building 615. Also, a pair of quality control and quality assurance samples was taken from sample location HG-04, at 3 feet and 6 feet above the ground. The air samples were collected using sorbent air sample tubes connected to low-flow vacuum air pumps, over a period of approximately eight hours. A total of 26 air samples (and one field blank) were collected. The sampling locations for the laboratory analyzed mercury sample locations are shown in Figure 2-3a and 2-3b.

2.3.3 Building 615 Drainpipe

During the site visit to perform the indoor air sampling a floor drain was located outside the photography laboratory, just inside the front door of Building 615. First, a visual inspection showed an upper steel pipe about 1 to 1.5 inches in diameter, which led in the direction of the Building 615 mechanical areas. A close evaluation determined that small, shiny, drops appeared to be inside the upper pipe. Below it was a larger diameter pipe which led in an unknown direction (see photograph in Appendix I, Page 12). To gain access to the lower pipe, a portion of the upper pipe had to be cut off. The piece that was removed was placed in a double plastic bag and stored on ice to be shipped to the lab for mercury analysis, as the first sediment sample (on August 4, 2004). An attempt was made to investigate the lower pipe by flushing 8-15 gallons of dyed water down the drain (see Appendix H).

A bore-o-scope and manual plumbers snake was "pushed" through the mud like material in the pipe. Eventually, the pipe ended in a tank or vault or drywell under the parking lot, just outside Building 615's photography lab. The beginning of this vault/drywell was approximately 10 feet outside the photography laboratory and 15 to 20 feet inside the pipe. The second sediment sample was collected at the pipe and vault/drywell junction on August 30, 2004. Figure 2-4 shows the locations of the two drainpipe samples.

3.0 SAMPLING RESULTS AND DATA EVALUATION

All the samples were collected in accordance with the Work Plan (USACE, 2004) and shipped by overnight express to their respective laboratories at the end of the sampling day. A copy of the chain of custody form showing proper turnover to the laboratory was returned to the Project Chemist at the completion of the project. The primary laboratory for the soil and groundwater samples analysis is TriMatrix Laboratories, Inc. Most of the QA samples analysis for the same media was performed by Northeast Analytical laboratory. The exception is that New York State Laboratory analyzed the QA samples from the resampling of soil boring SS-B11 on August 25, 2004. (See Section 3.1, for a fuller explanation.)

Sample chain of custody records show that all the soil and groundwater samples arrived within analysis holding times and in good condition (cool enough and unbroken/not leaking). All but one of the air samples (QA sample HG-04-H), arrived in good condition. The primary laboratory for the air samples analysis was Adirondack Environmental Services and the QA for air samples was performed by Galson Laboratories. A complete review of data quality is provided in Appendix G, the Data Usability Summary Report and the Chemical Quality Assurance Report.

3.1 Soil Sampling Results

The eleven soil borings collected had two samples each (shallow: 2 – 12 inches bgs and deep: 12 – 24 inches bgs). There were also four QC and two QA samples analyzed for metals and four QC and one QA samples analyzed for SVOCs. The results are in Tables 3-1 and 3-2, for SVOCs and metals, respectively. Sampling location SS-B11 was resampled on August 25, 2004 because of the wide range of SVOC values between primary, duplicate and quality assurance (QA) samples at the shallow depth, when the initial samples were analyzed.

Of the four metals identified for further investigation in the draft RI (e.g., arsenic, cadmium, chromium and mercury), arsenic and cadmium were seldom detected above New York TAGM levels. Arsenic concentrations ranged from 2.1 mg/kg to 10.0 mg/kg. Cadmium values ranged from 0.047 (J) mg/kg to 2.6 mg/kg. Chromium was present at a concentrations greater than the TAGM value of 10 mg/kg, in every soil boring, ranging from 15.0 mg/kg to 33.5 mg/kg. However the chromium TAGM is “10 mg/kg or soil background”, and soil background levels have not been established. There were several detections of mercury above NYSDEC’s TAGM of 0.1 mg/kg. Mercury values ranged from 0.0077 (J) mg/kg to 2.7 mg/kg. The significance of the metal and SVOC concentrations that exceeded TAGM values is addressed in the risk assessment.

Many of the metals results have laboratory data qualifiers (e.g., UJ) due to the low concentrations of present in the soil relative to the analytical detection limit. In addition, many of the TAGM concentrations are near the practical quantitation limits of the analytical methods selected. A list of laboratory qualifiers and their definitions is included with the tables for metals and organics.

The only samples that didn’t have at least one SVOC present at a concentration that was greater than the TAGM concentration were SS-3-DP, SS-4-DP, SS-5-DP, SS-9-DP, SS-10-DP, and SS-9-SH and its QC duplicate. Many of the SVOC results have laboratory data qualifiers (e.g., UJ) due to the low concentrations of present in the soil relative to the analytical detection limit. In addition, many of the TAGM levels are near the practical quantitation limits of the analytical methods selected. A noticeable observation is that the concentration of SVOCs tended to be lower at the 12 – 24 inch bgs interval than in the shallower (2 – 12 inches bgs) samples.

Sampling location SS-B11 was resampled because of the wide range of SVOC concentrations between the primary, duplicate and quality assurance (QA) samples at the shallow depth. Observation of the sample revealed that the soil was laden with coal, coal ash, asphalt, glass, and other debris. Despite the attempt at homogenizing the sample, the previous sample had not been sufficiently homogenized. The confirmation sample, analyzed August 26, 2004, showed similar concentrations of SVOCs in the primary, duplicate, QA and resampled results.

Of the chemicals identified and evaluated in the draft RI report, lead and PAHs were of specific concern to the NYDEC and NYDOH. The supplemental sampling results for lead showed a slight increase in the overall lead concentrations and no change in the overall concentrations of PAHs. The initial and supplemental sampling results for lead are shown in Figure 3-1. The initial and supplemental sampling results for total PAHs Figure 3-2.

3.2 Groundwater Results

The groundwater sample was analyzed for SVOCs in accordance with the Work Plan (USACE, 2004) because they had been detected in the previous samplings from MW-4. Figure 3-3 shows the locations of all of the monitoring wells on the FUDS property. The concentrations of SVOCs in the groundwater sample from MW-4R do not exceed the TOGS promulgated value for any compound. However, there were SVOCs in the groundwater from MW-4R which indicates that the source of SVOCs in MW-4 was not only from the parking lot runoff as previously suspected. The source of the SVOCs in the groundwater is unknown but could be from historical fill material used to level the area along the seawall. The salinity of Little Bay is approximately 20 parts per thousand, and potential intrusion of Little Bay surface water into the groundwater near MW-4R may be sufficient to make the groundwater undrinkable. The groundwater results are presented in Table 3-3.

3.3 Air Monitoring for Mercury in Building 615

Real time air monitoring samples from Building 615 range from 0.006 $\mu g/m^3$ to 0.038 $\mu g/m^3$, for the sample 1-inch above the drain in front of the photo laboratory room. The drainpipe sample inside the pipe registered 0.082 $\mu g/m^3$. The background air samples had mercury detections of 0.003 $\mu g/m^3$ to 0.006 $\mu g/m^3$.

The laboratory air samples did not contain mercury at a concentration greater than the instrument detection limit, thus every sample has a "U" data qualifier. The range of the nondetected results of the laboratory analyzed mercury air samples was from 0.54 $\mu g/m^3$ to 1.3 $\mu g/m^3$. The sample with the highest nondetected mercury concentration, sample HG-10-H (sample 10 at 6 feet above the floor), is above the state screening level of 1.0 ug/m^3. On sample HG-10-H, the sampling pump failed about half way through the sampling period, when only about 19.2 liters of air had been drawn through the absorbent tube. (The standard amount of air to pass through the sorbent tubes is between 40 and 45 liters.) The smaller sampling period and lower sample volume resulted in an elevated detection limit of 1.3 $\mu g/m^3$. Sample tube HG-04-H, a laboratory QA sample, was broken before it got to the laboratory. The mercury results are presented in Tables 3-4 and 3-5.

For perspective, the current Occupational Safety and Health Administration (OSHA) permissible exposure limit (PEL) for mercury vapor is 100 ug/m^3 of air as a ceiling limit. (A worker's exposure should never exceed ceiling limits.) The National Institute for Occupational Safety and Health (NIOSH) has established a recommended exposure limit (REL) for mercury vapor of 50 ug/m^3, as a TWA for up to a 10-hour workday and a 40-hour workweek. The American Conference of Governmental Industrial Hygienists (ACGIH) has assigned mercury vapor a threshold limit value (TLV) of 25 ug/m^3, as a TWA for a normal 8-hour workday and a 40-hour workweek. ACGIH and NIOSH note that mercury vapor absorption through the skin, mucous membranes and eyes, contributes to overall exposure.

3.4 Building 615 Drainpipe, Sludge Mercury Sample Results

The first drainpipe sludge material sample, collected August 4, 2004, had a mercury concentration of 6.25 mg/kg. The second drainpipe sludge material sample, collected August 30, 2004, had a mercury concentration of 10.0 mg/kg. The results are presented in Table 3-6.

The extent of the lower drain pipe was investigated by flushing 8-15 gallons of dyed water down the drain (see Appendix H). After flushing the dye liquid down the drain, a search of the water side of the seawall did not indicate any outfalls into Little Bay.

4.0 HUMAN HEALTH RISK ASSESSMENT FOR UPLAND AREA

The purpose of this supplemental human health risk assessment (HHRA) was to determine whether there are potential human health risks associated with constituents of potential concern (COPCs) in environmental media in upland areas of the Fort Totten Coast Guard Station. The HHRA for the Coast Guard Station was conducted in accordance with USEPA's *Risk Assessment Guidance for Superfund (RAGS) Part A* (USEPA 1989) and with the USACE guidance document *Risk Assessment Handbook Human Health Evaluation, Volume I: Human Health Evaluation* (USACE 1995), and incorporates data collected in 2004 in addition to the data previously reported in the original RI (USACE 2002). Exposure and toxicity assumptions presented in USACE (2002) have been utilized for this supplemental HHRA.

4.1 Introduction

Data for this evaluation were taken at different sampling periods. Data from the first Site Investigation (SI) have not been included in this risk assessment (USACE 1988). The 1988 SI data were not used because the data were not validated in accordance with USEPA requirements. Data from samples taken in 1998 during Phase 1 and 2000 during Phase 2 were combined and used for this HHRA. Monitoring wells were resampled in April 2002, and these data have been included in this HHRA. Additional surface soil samples were taken in 2004 from what are classified as the Fill Area (2 extra samples) and Other Area (9 extra samples) from 2-12 inches below surface level and 12-24 inches below surface level. Metals and SVOCs were analyzed in these new 2004 soil samples. These data have been incorporated into the existing data for the assessment of risks to human receptors in the Fill and Other Areas. Monitoring well MW-4 had been compromised, therefore the existing well was decommissioned and a replacement developed and sampled. Only SVOCs were analyzed in this replacement well. In addition, real-time and time-integrated measurements of mercury in air were taken in Building 615 using an instrument capable of achieving detection limits lower than a screening value of 1 $\mu g/m^3$ (NYSDEC 2004). Detailed discussions regarding the sampling events are provided in Chapter 2.

Data have been characterized according to the locations associated with these samples. Areas that have been assessed independently for total soil include:

- Fill Area
- Other Area (e.g., all other soil samples)

These areas are shown in Figure 4-1. Risks from groundwater were characterized as one exposure unit across the entire site. The results of the air monitoring of mercury in Building 615 are also addressed separately.

The samples associated with each of these areas which were included in the risk assessment are shown in Table 4-1.

The risk assessment methodology used in this HHRA involves the same a four-step process used in the draft RI report (USACE, 2002): hazard identification, exposure assessment, toxicity assessment, and risk characterization.

4.2 Hazard Identification

A hazard identification was conducted to determine which constituents are of potential concern at the site. Typically in the hazard identification, site-specific data are analyzed and compared to risk-based screening values; however, for this risk assessment no screening was performed, and with the exception of essential nutrients, all chemicals detected in each of the matrix/location areas were carried through the risk assessment.

4.2.1 Conceptual Site Model and Identification of Potential Exposure Pathways

A conceptual site model was developed for the upland areas of the Fort Totten Coast Guard Station to depict the potential pathways of concern at the site and is provided in Figure 4-2 for soil and Figure 4-3 for groundwater.

4.2.1.1 Media of Concern

Media of concern included total soil (combined surface and subsurface soil) and groundwater as environmental transport media for the release of chemicals present in the upland areas of the Fort Totten Coast Guard Station.

4.2.1.2 Exposure Pathways and Receptors of Concern

An exposure pathway describes a mechanism by which a population or individual may be exposed to chemicals present at a site. A completed exposure pathway requires the following four components:

- A source and mechanism of chemical release to the environment;
- An environmental transport medium for the released chemical;
- A point of potential human contact with the contaminated medium; and
- A human exposure route at the point of exposure.

All four components must exist for an exposure pathway to be complete and for exposure to occur. Incomplete exposure pathways do not result in actual human exposure and are not included in the exposure assessment and resulting risk characterization.

The April 1998 final reuse plan (Fort Totten Redevelopment Authority 1998) proposed that the area remain in public ownership with uses as a publicly accessible park. Consequently, one category of the current and future receptors includes recreational adults and children. Children are much more sensitive receptors than adults; therefore, soil risks will be quantified for recreational adolescents (ages 6-15) (Figure 4-2). Adult recreational risks will be qualitatively addressed. Although the expected future land use scenario is recreational, an analysis was conducted to account for un-restricted use (i.e., residential re-use) by examining total soil risks for residential adults and children as well as commercial workers (Figure 4-2). In addition, it is possible that residential adults and children may be exposed to contaminants through the ingestion of homegrown fruits and vegetables. These risks will be quantified, and included in the other exposure pathways shown in Figure 4-2. Because it would be necessary to do construction for either of these potential future uses, risks from exposure to total soil for construction workers will be quantified (Figure 4-2). Finally, to be consistent with New York State requirements, risks for groundwater consumption and bathing by residential adults and children will be quantified, even though the groundwater is not presently used (Figure 4-3). It is unlikely that new residences would use the groundwater; rather, they likely would be connected to the public water system. For these reasons, it is unlikely that groundwater will be used by potential future residential adults and children. Because carcinogenic risks are evaluated on a lifetime risk basis, the residential adult and child calculated risks are combined to account for potential lifetime residential exposure to the site.

As a conservative measure, current and future recreational adolescents were evaluated for potential risks associated with incidental ingestion of, dermal contact with, and inhalation of particulates entrained from total soil (surface and subsurface), even though exposure to subsurface soil is not expected for this receptor. The use of total soil as the exposure medium is due to construction activities that may mix existing subsurface soil with surface soil.

Future resident adults and children were evaluated for potential risks associated with incidental ingestion of, dermal contact with, and inhalation of particulates entrained from total soil. Future residential adults and children will also be quantitatively assessed for the consumption of homegrown fruits and vegetables. Assessing residential risks for total soil, which assumes contact with both surface and subsurface soil, is a conservative estimate to ensure that site soil is evaluated for any potential future use.

Future commercial workers were evaluated for incidental ingestion of, dermal contact with, and inhalation of particulates from total soil. Commercial workers typically are not involved in digging scenarios where exposure to subsurface soil would occur; however, assessment of total soil risks represents a conservative estimate of exposure. As with residents, total soil was used as the exposure medium as a conservative measure to account for potential mixing of subsurface with surface soil as a result of construction activities.

Future construction workers are evaluated for incidental ingestion of, dermal contact with, and inhalation of particulates from total soil during excavation activities. It was assumed that construction workers would contact both surface and subsurface soil; therefore, exposure to total soil is evaluated.

The aquifer under the Ft. Totten FUDS is not currently used as a water source. The salinity of Little Bay is approximately 20 parts per thousand, and potential intrusion of Little Bay surface water into the groundwater may be sufficient to make the groundwater undrinkable. However, future residential use of groundwater at the site was evaluated as a conservative measure. Future residential adult and child exposures to groundwater via tap water through ingestion and dermal pathways were assessed. Inhalation is a probable pathway of concern for groundwater when there are VOCs of concern in groundwater. Because VOCs were detected in groundwater, inhalation of volatile constituents while showering was assessed for the adult resident via shower model by Foster and Chrostowski (1987). It was assumed that children (up to age 6) would be bathing, and volatile inhalation is not a significant pathway of volatiles for this receptor.

4.2.2 COPCs Selected

Under most circumstances, analytes detected in media of concern such as soil or groundwater are screened using risk-based toxicity values; however, for this HHRA, if metals or chemicals were detected in a medium, they were considered to be COPCs, and risks were quantified for them. The exception to this is that risks for essential nutrients (calcium, iron, magnesium, potassium, and sodium) have not been quantified due to their inherent importance for human health and their relatively low toxicity.

Analytes detected in total soil from the Fill Area are summarized in Table 4-2. Once the five essential nutrients were removed, 18 metals were detected and require quantitative risk assessment in this area. In addition, 17 PAHs, 4 SVOCs, and 7 VOCs were detected and will be quantitatively assessed. Similarly, 18 metals, 17 PAHs, 8 SVOCs, and 7 VOCs will require quantitative risk characterization in the Other Area (Table 4-3).

Mercury was never detected in Buiding 615 indoor air at the concentration greater than the detection limit. For all but one sample the detection limits were below the screening value of 1 μg/m^3. The pump failed approximately half way through the normal sampling time for Sample AD2004-AIR-HG-10-H resulting in a detection limit of 1.3 μg/m^3. However, in accordance with USEPA risk assessment guidance (USEPA 1989), when a chemical is not detected to use ½ of the detection limit as a surrogate concentration. In that situation, this sample also resulted in a mercury concentration below the 1 μg/m^3. Consequently, risks from the inhalation of mercury in Building 615 have not been quantified.

Finally 18 nonessential metals, 15 PAHs, 7 SVOCs, 1 pesticide and 13 VOCs were detected at least once in groundwater (Table 4-4) and will be quantitatively assessed for risk to residential adults and children.

4.3 Exposure Assessment

An exposure assessment is conducted to estimate the magnitude of potential human exposures to COPCs in site media. Typically in exposure assessment, average and reasonable maximum estimates (RME) of potential exposure are developed in accordance with USEPA guidance for both current and potential future land-use assumptions. However, average exposure estimates are rarely utilized for HHRA, and consequently only RME risks are presented in this risk assessment. Conducting an exposure assessment involves analyzing releases of COPCs; identifying all potential pathways of exposure; estimating RME exposure point concentrations for specific pathways, based both on environmental monitoring data and predictive chemical modeling results; and estimating potential chronic daily intakes for specific pathways. The results of this assessment are pathway-specific estimates of potential intakes for current and future exposures to individual chemicals of potential concern.

4.3.1 Quantification of Potential Exposures

The first step of the exposure assessment is to quantify potential exposure concentrations. This involves the evaluation of site data and the quantification of exposure concentrations for RME exposure scenarios.

4.3.1.1 Data Quality Evaluation

Inclusion or exclusion of data on the basis of analytical qualifiers was performed in accordance with USEPA guidance (USEPA 1989). Highlights related to the HHRA are presented here:

- Analytical results bearing the U qualifier (indicating that the analyte was not detected at the given sample quantitation level) were retained in the data set and considered non-detects. Where warranted for statistical purposes, each COPC was assigned a numerical value of one-half of the reported detection limit.

- Analytical results rejected were assigned an R qualifier and were not included in the risk assessment.

- Analytical results bearing the J qualifier [indicating that the reported value was estimated because the analyte was detected at a concentration below the sample quantitation limit (SQL) or for other reasons] were retained at the reported concentration.

- Analytical results showing any other qualifiers (B, N, *, P, D, E-see Tables 4-2 through 4-4) were retained at the reported concentration.

- Concentrations from duplicate samples were averaged to determine the appropriate concentration for that specific sample.

- Common laboratory contaminants, including acetone, 2-butanone, methylene chloride, chloroform, toluene, phthalate esters, and uncommon laboratory contaminants were considered to be COPCs unless it was evident that their presence was not related to site-specific activities but were due to laboratory contamination.

4.3.1.2 Estimation of RME Concentrations

To assess human health risks, a statistical analysis of the COPC concentrations in each medium was performed. The methods used to analyze the data for each of these media are described below.

Total soil and groundwater are potential site media of concern. For total soil and groundwater, reported concentrations were used to calculate the 95 upper confidence limit of the mean (95 UCLM) for COPCs in each medium (USEPA 1992b). Exposure point concentrations (EPCs) in site media were estimated as the 95 UCLM values for purposes of estimating the RME. In cases where the 95 UCLM values exceeded the maximum detected concentration, the maximum detected concentration was used.

The first step in estimation of EPC was to determine whether medium-specific environmental data for a COPC were normally or log-normally distributed. This was accomplished with the Shapiro-Wilks W-test for distribution (Gilbert 1987). The distribution was determined by comparison of the calculated W-statistic with critical W-statistic values. If the distribution fit neither normal or log-normal, a standard bootstrap method was used to estimate the 95 UCLM. Analysis using the standard bootstrap method is further discussed below.

If the statistical test supported the assumption that the data set for a COPC was normally distributed the following steps were undertaken to calculate 95 UCLM (USEPA 1992a): (1) calculate the arithmetic mean of the untransformed data; (2) calculate standard deviation of the untransformed data; (3) determine the one-tailed t-statistic (Gilbert 1987); and (4) calculate 95 UCLM.

For a log-normally distributed COPC, the following steps were performed to calculate 95 UCLM. Because transformation is a necessary step in calculating the UCLM for a log-normal distribution, the data were transformed by using the natural logarithm function (i.e., calculate ln(x), where x is the value from the data set). After transforming the data, 95 UCLM for the data set was found by calculating the arithmetic mean of the transformed data; calculating standard deviation of the transformed data; determining H-statistic (Gilbert 1987); and calculating 95 UCLM.

For cases where the distribution of analyte concentrations deviated strongly from both normal and log-normal distributions, the 95 UCLM was determined using standard bootstrap estimation. The bootstrap approach (Efron 1981) was used to estimate the standard error of a sample statistic, $\hat{\theta}$, without making any assumptions of how the original data are distributed. The following is a brief description of how the standard bootstrap procedure is used to estimate the upper confidence limit (UCL) of a sample statistic (Singh et al. 1997).

Step 1. From the original sample $\boldsymbol{X_n} = (X_1, X_2, \ldots, X_n)$; where the deviates X_i are independently and identically distributed, draw a sample of n observations with replacement such that each observation has the same probability of being drawn ($= \frac{1}{n}$). The new data set is called the bootstrap sample, and is typically denoted as $\boldsymbol{X_n^*} = (X_1^*, X_2^*, \ldots, X_n^*)$.

Step 2. Compute the sample statistic, $\hat{\theta}^*$, of interest (in this case the sample mean $\bar{X}$) from $\boldsymbol{X_n^*}$.

Step 3. The procedures in Steps 1 and 2 are repeated 2,000 times generating 2,000 bootstrap estimates of the sample statistic. The general bootstrap estimate is the arithmetic mean of the 2,000 estimates, $\bar{\theta}_B = \frac{1}{2000}\sum_{i=1}^{2000}\hat{\theta}_i^*$. The bootstrapped standard error of $\hat{\theta}$, denoted by $\hat{\sigma}_B$, is given by

$$\hat{\sigma}_B = \sqrt{\frac{1}{2000-1}\sum_{i=1}^{2000}\left(\hat{\theta}_i - \bar{\theta}_B\right)^2}\,.$$

Step 4. Finally, the (1-p)100% confidence limits of $\hat{\theta}$ are given by

$$\hat{\theta} \pm z_p \hat{\sigma}_B.$$

where z_p is the p^{th} quantile of the standard normal distribution.

Tables 4-5 and 4-6 show the RME EPC for each COPC in total soil for the Fill Area and Other Area respectively. Table 4-7 summarizes the RME EPC for groundwater COPCs. The RME EPC value was utilized as the chemical-specific, medium-specific EPC in the exposure assessment for the risk assessment.

4.3.2 Exposure Equations

The next step in the exposure assessment was to estimate COPC intakes for each of the pathways considered in the assessment. In this exposure assessment, two different measures of intake, depending on the nature of the effect being evaluated, are provided. When evaluating longer-term (i.e., subchronic and chronic) exposures to chemicals that produce adverse non-carcinogenic effects, intakes are averaged over the period of exposure (i.e., the averaging time [AT]) (USEPA 1989). This measure of intake is referred to as the non-carcinogenic average daily intake (ADI) and is a less than lifetime exposure. For chemicals that produce carcinogenic effects, intakes are averaged over an entire lifetime and are referred to as the lifetime average daily intake (LADI) (USEPA 1989).

The generic equation to calculate intakes is given below:

$$(L)ADI = \frac{C \; x \; IF \; x \; EF \; x \; ED \; x \; RAF}{BW \; x \; AT} \text{ x CF}$$

where:

(L)ADI	=	(Lifetime) Average daily intake (mg/kg-day)
C	=	Concentration in a specific medium (mg/L or mg/kg)
IF	=	Intake factor[1] (mg/day)

[1] The intake factor is the product of all intake variables that, when multiplied by the concentration of the chemical of potential concern in a specific medium, results in an estimate of the chemical intake in mg/kg-day for that population and exposure pathway. Intake factors may include ingestion rate, inhalation rate, body surface area exposed to soil or water, dermal permeability constants, and soil adherence factors.

EF	=	Exposure frequency (days/year)
ED	=	Exposure duration (years)
RAF	=	Relative absorption factor (unitless)
BW	=	Body weight (kg)
AT	=	Averaging time (days)
CF	=	Conversion Factor (10^{-6} kg/mg)

The daily intake of COPC from ingested homegrown produce (I_{ag}) was calculated using the following equation (USEPA 1998c):

$$I_{ag} = [(\Pr_{ag} \times CR_{ag}) + (\Pr_{pp} \times CR_{pp}) + (\Pr_{bg} \times CR_{bg})] \times F_{ag}$$

where:

Pr_{ag}, Pr_{pp}, and Pr_{bg}	= COPC concentrations in aboveground produce, aboveground protected produce, and belowground produce respectively (mg/kg DW),
Cr_{ag}, CR_{pp}, and CR_{bg}	= consumption rate of aboveground produce, protected aboveground produce, and belowground produce respectively (kg/kg-day DW), and
F_{ag}	= fraction of produce consumed from contaminated area (unitless).

The dose from homegrown produce was calculated using:

$$(L)ADI = \frac{I_{ag} \times EF \times ED}{AT}$$

where all terms are defined as above.

4.3.3 Selection of Exposure Factor Values

All exposure factor values used in estimating intakes are described and referenced in Tables 4-8 to 4-17. Various USEPA guidance documents were used in defining exposure factor values for estimating intakes for exposure pathways evaluated at upland areas of the Fort Totten FUDS (e.g., USEPA 1989, USEPA 1991, USEPA 1997d, USEPA 1992a, USEPA 2000a, and USEPA 1998c).

For all exposure pathways that have exposure factor values specified in RAGS Part A and in OSWER Directive 9285.6-03, those values were used in this risk assessment. Dermal Guidance documents from USEPA (1992a and 2000a) were also utilized. All exposure factor values utilized are presented in the following sections.

4.3.3.1 Future Residents

Future residential users may potentially be exposed to COPCs via total soil and groundwater. Both adults and children were assessed for the residential scenario. Cancer risks were assessed based on combined child/adult risks for a total exposure of 30 years. Non-cancer risks for residential adults were based on a 30-year exposure and for children on a 6-year exposure.

Residential Adults – Exposure parameters for residential adult exposure are presented in Table 4-8 for total soil and Table 4-9 for groundwater. Body weight for the adult resident was assumed to be 70 kg. Under RME conditions, future adult residents were assumed to have an exposure duration of 30 years for non-carcinogenic hazards and an exposure duration of 24 years for carcinogenic risks. Both carcinogenic and non-carcinogenic risks had an exposure frequency of 350 days/year. Surface area available for dermal exposure to soil was assumed to be 5,700 cm2. An adherence factor of 0.07 mg/cm2 for soil was assumed (USEPA 2000a). An inhalation rate of 0.83 m3/hr for 24 hr/day for soil was conservatively assumed (Table 4-8). The residential adult RME groundwater ingestion rate was assumed to be 2 liters per day (Table 4-9). Other standard groundwater exposure parameters were used for ingestion and dermal contact, and are shown in Table 4-9. Surface area exposed to groundwater while bathing/showering was assumed to be 18,000 cm2 for 0.58 hour (35 minutes) exposure time (Table 4-9). Exposure values used to assess volatilization and inhalation of volatile organic chemicals while showering are shown in Table 4-10. Consistent with the upper limit of exposure, a 15-minute shower for 350 days/year for 30 years was assumed for this risk assessment (Table 4-10). Exposure constants for the adult residential consumption of homegrown vegetables are shown in Table 4-11. Consistent with combustion guidance (USEPA 1998c), different consumption rates were used for aboveground produce, belowground produce (carrots or potatoes), and aboveground protected produce (vegetables with a skin that is removed such as cucumbers). Also consistent with the combustion guidance, it was assumed that 25 percent of consumed produce was contaminated. Finally, it was assumed that this exposure would occur for 2 months out of the year (60 days) (Table 4-11).

Residential Children – Exposure parameters for child resident exposure are presented in Table 4-12 for total soil and Table 4-13 for groundwater. Body weight for the future child resident was assumed to be 15 kg. Under RME conditions, future child residents were assumed to have a 6-year exposure duration with an exposure frequency of 350 days/year. RME surface area available for dermal exposure to soil was assumed to be 2,800 cm2 with an adherence factor of 0.2 (USEPA 2000a). An inhalation rate of 0.417 m3/hr for 24 hr/day was assumed (USEPA 1991). The residential child RME groundwater ingestion rate was assumed to be 1 liter per day (Table 4-12). RME surface area exposed to groundwater while bathing was assumed to be 6,600 cm2 for 1.0 hour exposure time. As noted in the exposure pathway discussion, inhalation of volatiles by children during bathing was assumed to be *de minimus*, and therefore risks for this exposure scenario were not quantified. Exposure constants for the child residential consumption of homegrown vegetables are shown in Table 4-14. Consistent with combustion guidance (USEPA 1998c), different consumption rates were used for aboveground produce, belowground produce (carrots or potatoes), and aboveground protected produce (vegetables with a skin that is removed such as cucumbers). Also consistent with the combustion guidance, it was assumed that 25 percent of consumed produce was contaminated. Finally, it was assumed that this exposure would occur for 2 months out of the year (60 days) (Table 4-14).

4.3.3.2 Current and Future Adolescent Recreational User

Exposure parameters for adolescent recreational user exposure are presented in Table 4-15 for total soil. The age range of the adolescent recreational user was assumed to be 6 to 15 years, and the body weight for the adolescent recreational user was assumed to be 36 kg. Under RME conditions, adolescent recreational users were assumed to have an exposure duration of 9 years with an exposure frequency of 141 days/year. This exposure frequency is based on an average of the mean days of outdoor activity for a young child (130 days/year) and an older child (152 days/year) (USEPA 1997d). A surface soil ingestion rate of 200 mg/day was assumed. RME dermal exposure to soil was based on 2,900-cm^2 surface area. The dermal adherence factor was assumed to be 0.3 mg/cm^2 based on teenaged soccer players exposed to moist soil (USEPA 2000a). The RME inhalation rate was assumed to be 0.83 m^3/hr for particulates over an exposure time of 10 hr/day.

4.3.3.3 Construction Worker

Total soil exposure parameters for construction worker exposure are presented in Table 4-16. Body weight for the construction worker was assumed to be 70 kg. Under RME conditions, future construction workers were assumed to have an exposure duration of 1 year with an exposure frequency of 150 days/year. The exposure frequency was determined by assuming the construction worker would be at the site 5 days a week for 30 weeks. Skin surface area available for contact with total soil during construction activities was assumed to be 3,300 cm^2. Incidental ingestion of soil was assumed to be 480 mg/day (USEPA 1997d). The RME inhalation rate was assumed to be 0.83 m^3/hr for total soil particulate over an exposure time of 8 hr/day.

4.3.3.4 Commercial Worker

Exposure parameters for commercial worker exposure are presented in Table 4-17 for total soil. Body weight for the commercial worker was assumed to be 70 kg. Under RME conditions, commercial workers were assumed to have an exposure duration of 25 years with an exposure frequency of 250 days/year. A soil ingestion rate of 50 mg/day was assumed. Dermal exposure to soil was based on 5,700 cm^2 surface area based on a residential adult and 0.1 mg/cm^2 adherence factor based on groundskeepers (USEPA 2000a). The RME inhalation rate was assumed to be 0.83 m^3/hr for particulates over an exposure time of 8 hr/day.

4.3.3.5 Modeling Contaminant Concentrations in Homegrown Produce

Concentrations of COPCs in homegrown produce were estimated based on models found in the literature (USEPA 1998c, Baes et al. 1984, Bechtel Jacobs 1998, and Travis and Arms 1988). The order in which the different models were used was:

1. USEPA (1998c) is guidance provided by the Center for Combustion Science and Engineering. Equations presented in Appendix Tables B-2-9 and B-2-10 of this citation were used to estimate COPC concentrations in aboveground and belowground produce respectively. Chemical specific parameters used in these equations were taken from Appendix A-3 of this guidance.

2. In the absence of chemical-specific parameters in USEPA (1998c), the next reference used to estimate produce concentrations was Bechtel Jacobs (1998). This guidance from Oak Ridge lists regression equations to estimate vegetation concentrations relative to soil concentrations. This was only used to calculate copper produce concentrations.

3. In the absence of accumulation factors in the above two references, Baes et al. (1984) was used for metals (aluminum, cobalt, manganese, and vanadium) and Travis and Arms (1988) was used for organic chemicals (acenaphthylene, benzo(g,h,i)perylene, carbazole, and dibenzofuran).

The details of equations used for these various sources are discussed below, and chemical-specific parameters for each COPC are shown in Table 4-18.

The combustion guidance (USEPA 1998c) provides equations for the calculation of aboveground produce and belowground produce. Aboveground produce contaminant concentrations are calculated using the following equation:

$$\mathrm{Pr}_{ag} = Cs \times Br_{ag}$$

where:

Pr_{ag} = Concentration of COPC in aboveground produce due to root uptake (mg/kg DW),
C_s = Concentration of COPC in soil (mg/kg DW), and
Br_{ag} = Plant-soil bioconcentration factor for aboveground produce.

Belowground produce concentrations were calculated using the following equation:

$$\Pr_{bg} = Cs \times Br_{rootveg} \times VG_{rootveg}$$

where:

Prbg = Concentration of COPC in belowground produce due to root uptake (mg/kg DW),

Brrootveg = Plant-soil bioconcentration factor for belowground produce (unitless), and

Vgrootveg = Empirical correction factor for belowground produce (unitless).

For the uptake of copper into plants, Bechtel Jacobs (1998) used to estimate copper concentrations. This source derived a log-log regression equation directly relating expected concentrations in plants relative to concentrations in the soil:

$$\ln[X]_{plant} = B_0 + B_1\ x \ln[X]_{soil}$$

where:

$\ln[X]_{plant}$ = the natural log of the concentration of chemical X in earthworm, plant or small mammal (mg/kg DW),
B_0 and B_1 are constants identified in Bechtel Jacobs (1998) for plants (0.669 and 0.394 for copper, respectively),
and $\ln[X]_{soil}$ = the natural log of the concentration of copper in soil (mg/kg DW).

The inverse of the natural log plant concentration was then taken, and used as the estimated copper concentration in plants.

For the metals that were not addressed in the Combustion Guidance (USEPA 1998c) Baes et al. (1984) was used to estimate concentrations of selected COPCs (aluminum, cobalt, manganese, and vanadium) in aboveground vegetation and protected aboveground vegetation (cucumbers, etc.). These authors examined the uptake of metals into these parts of vegetation, and derived bioconcentration factors.

For the quantification of metals in aboveground vegetation:

$$C_v = B_v \times C_s$$

where:

C_v = concentration of metal in aboveground produce (mg/kg dry),

B_v = soil to plant bioconcentration factor for vegetative portions of the plant (unitless), and

C_s = concentration of metal in soil (mg/kg dry).

For the quantification of metals in aboveground protected vegetation:

$$C_r = B_r \times C_s$$

where:

C_r = concentration of metal in aboveground protected produce (mg/kg dry), and

B_r = soil to plant bioconcentration factor for protected aboveground portions of the plant (unitless),

For organic chemicals not specifically addressed in the combustion guidance (USEPA 1998c), a regression equation derived by Travis and Arms (1988), based on the chemical's log K_{ow}, was used to estimate produce concentrations:

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

where log B_v = the base 10 log value of the bioconcentration factor, and log K_{ow} is the chemical's physical property.

If any specific type of produce concentration was not calculable (for example belowground produce from Baes equations or protected aboveground concentrations from combustion guidance calculations), concentrations from aboveground produce were utilized for exposure concentrations.

Constants used in the above equations are shown in Table 4-18 and predicted concentrations of aboveground, belowground, and protected produce are shown in Table 4-19.

4.4 Toxicity Assessment

The toxicity assessment considers the types of potential adverse health affects associated with exposures to COPCs; the relationship between magnitude of exposure and potential adverse effects; and related uncertainties, such as the weight of evidence of a particular COPC's carcinogenicity in humans. The toxicity assessment for COPCs relies on existing toxicity information developed on specific organic compounds and inorganic constituents. USEPA Guidance (USEPA 1989) specifies that the assessment is accomplished in two steps: hazard identification and dose-response assessment. Hazard identification is the process of determining whether studies claim that exposure to a COPC may cause the incidence of an adverse effect. USEPA specifies the dose-response assessment, which involves: (1) USEPA's quantitative evaluation of the existing toxicity information, and (2) USEPA's characterization of the relationship between the dose of the COPC administered or received, and the incidence of potentially adverse health effects in the exposed population. From this quantitative dose-response relationship, specific toxicity values are derived by USEPA that can be used to estimate the incidence of potentially adverse effects occurring in humans at different exposure levels (USEPA 1989). These USEPA-derived toxicity values are called Reference Doses (RfDs) for non-carcinogens and Slope Factors (SFs) for potential carcinogens. The toxicity values used for COPCs at the Fort Totten Coast Guard Station are presented in Tables 4-20 and 4-21 for non-carcinogens and in Tables 4-22 and 4-23 for carcinogens. Chemical-specific parameters including absorption factors (ABS) and permeability constants are shown in Table 4-24.

4.4.1 Toxicity Assessment for Non-Carcinogens

For most COPCs, toxicity values for non-carcinogens were taken, when available, from the Integrated Risk Information Systems (IRIS) database (USEPA 2001). IRIS chronic toxic potency concentrations are developed by USEPA and undergo an extensive process of scientific peer review. Therefore, IRIS values are judged to be adequately verified.

If toxic potency concentrations for COPCs were not available from IRIS (USEPA 2001), Health Effects Assessment Summary Tables (HEAST) (USEPA 1997e) were used as a secondary data source. As HEAST toxicity values are not scientifically peer-reviewed for quality or scientific acceptability, they may not be derived in strict accordance with USEPA-approved methodologies.

In the absence of toxicity data from IRIS or HEAST, toxicity data from the USEPA National Center for Environmental Assessment information (NCEA) was used for toxicity information.

If toxic potency concentrations were not available for one route of exposure, but existed for another route—for example, if an oral RfD existed but no inhalation RfD—the existing value was examined for technical applicability to the alternate route and subsequently utilized, if appropriate.

The methodology used by USEPA for deriving toxic potency concentrations for non-carcinogens, and site-specific considerations for modifying or using these concentrations are discussed in detail in Barnes and Dourson (1988) and USEPA guidance (USEPA 1989). Non-carcinogens are typically judged to have a threshold daily dose below which deleterious or harmful effects are unlikely to occur. This concentration is called the no-observed-adverse-effect-level (NOAEL) and may be derived from either animal laboratory experiments or human epidemiology investigations (usually workplace studies). In developing a toxicity value or human NOAEL for non-carcinogens (i.e., an RfD), the regulatory approach is first to (1) identify the critical toxic effect associated with chemical exposure (i.e., the most sensitive adverse effect); (2) identify the threshold dose in either an animal or human study; and (3) modify this dose to account for interspecies variability (where appropriate), differences in individual sensitivity (within-species variability), and other uncertainty and modifying factors. Uncertainty factors are intended to account for specific types of uncertainty inherent in extrapolation from the available data. Modifying factors account for the concentration of confidence in the scientific studies from which toxicity values are derived, according to such parameters as study quality and study reproducibility. The use of these factors is a conservative approach for protection of human health and is likely to overestimate the toxic potency associated with chemical exposure. The resulting RfD is expressed in units of milligrams of chemical per kilogram of body weight per day (mg/kg-bw/day).

Toxicity values used for exposures that involve dermal contact with chemicals typically require adjustment of the oral toxicity values (oral RfDs). This adjustment accounts for the difference between the daily intake dose through dermal contact as opposed to ingestion. Most toxicity values are based on the actual administered dose, and must be corrected for the percent of chemical-specific absorption that occurs across the gastrointestinal tract prior to their use in dermal contact risk assessment (USEPA 1989, 1992a and 2000a). USEPA (1998 and 2000a) recommend that dermal risks from PAHs not be addressed quantitatively, but rather that dermal risks from PAHs be discussed qualitatively. This is the approach taken for this HHRA. Dermal factors applied for this risk assessment are shown in Table 4-20 and 4-22 for non-cancer and cancer toxicity respectively.

4.4.2 Toxicity Assessment for Carcinogenicity

Unlike non-carcinogens, carcinogens are assumed to have no threshold. There is presumed to be no level of exposure below which carcinogenic effects will not manifest themselves. This "non-threshold" concept supports the idea that there are small, finite probabilities of inducing a carcinogenic response associated with every level of exposure to a potential carcinogen. The "no threshold" assumption is a science-policy decision, which is health protective, yet is not universally accepted within the scientific community. USEPA uses a two-part evaluation for carcinogenic effects. This evaluation includes the assignment of a weight-of-evidence classification and the quantification of a cancer toxic potency concentration. Quantification is expressed as a slope factor, which reflects the dose-response data for the carcinogenic endpoint(s) (USEPA 1989).

The weight-of-evidence classification system assigns a letter or alphanumeric (A through E) to each potential carcinogen that reflects an assessment of its potential to be a human carcinogen.[2]

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[2]A = a known human carcinogen; B1 = a probable human carcinogen, based on sufficient animal data and limited human data; B2 = a probable human carcinogen based on sufficient animal data and inadequate or no human data; C = a possible human carcinogen; D = not classifiable as to human carcinogenicity; and E = evidence of non-carcinogenicity for humans.

The weight-of-evidence classification is based on a thorough scientific examination of the body of available data. Only compounds that have a weight-of-evidence classification of A, B, or C are considered to have carcinogenic potential in this risk assessment. In 1996 USEPA published a document titled *Proposed Guidelines for Carcinogenic Risk Assessment* (USEPA 1996b). This document discusses changes in the USEPA risk estimation method away from tumor findings in animals and humans towards a more expanded approach that allows the incorporation of recent more sophisticated methods of assessing the carcinogenicity of a chemical. For example, if a chemical is found to be carcinogenic from only one route of exposure and not others, the method allows modification of the cancer slope factors to allow this in risk assessments. An example of this is the recent change in the cancer risk potential for beryllium, now focussing on the inhalation exposure route only and finding the element to be non-carcinogenic from other exposure pathways such as oral ingestion.

The USEPA slope factor is the upper 95th percentile confidence limit of the probability of response per unit daily intake of a chemical over a lifetime. Typically, the slope factor is used to estimate the upper-bound lifetime probability of a person developing cancer from exposure to a given concentration of a carcinogen. Slope factors are generally based on experimental animal data, unless suitable epidemiological studies are available. Due to the difficulty in detecting and measuring carcinogenic endpoints at low exposure concentrations, slope factors are typically developed by using a model to fit the available high-dose, experimental animal data, and then extrapolating downward to the low-dose range to which humans are typically exposed. USEPA usually employs the linear multistage model to derive a slope factor. The model is conservative, and provides an upper bound estimate of excess lifetime cancer risk. Thus, the actual risk may be lower and could be zero (USEPA 1989). These methods and approaches are discussed in greater detail in USEPA (1989).

Carcinogenic slope factors used for exposures that involve dermal contact typically require adjustment of the oral slope factor. This accounts for the difference between the dermal dose and the ingested dose. Most toxicity values are based upon the actual administered dose. The values must be corrected for the percent of chemical-specific absorption that occurs across the gastrointestinal tract prior to use in dermal contact risk assessments (USEPA 1989). As discussed above, USEPA (1989) has recommended a qualitative assessment of toxicity for PAHs, which has been done in this risk assessment. For inhalation exposures, inhalation slope factors are developed if sufficient data are available.

4.4.3 Toxicity of Constituents of Potential Concern

A review of relevant toxicity data for each COPC was performed using IRIS (USEPA 2001), a peer-reviewed toxicity database. If toxicity data were not found in IRIS toxicity data were taken from HEAST (USEPA 1997e), NCEA or ATSDR.

4.4.3.1 Summary of Toxicity Values for Non-Carcinogenic Effects

USEPA-derived toxicity values for evaluating potential chronic non-carcinogenic effects for COPCs are summarized in Tables 4-20 and 4-21. Toxicity information presented in these tables includes the following USEPA provided/derived information: chronic or subchronic RfD values for exposures via the oral and inhalation pathway, reported target organs, uncertainty and modifying factors specific to the USEPA-derived RfD, and the scientific source of the information.

4.4.3.2 Summary of Toxicity Values for Potential Carcinogenic Effects

USEPA-derived toxicity values for evaluating potential carcinogenic effects for COPCs are summarized in Tables 4-22 and 4-23. Toxicity information presented in these tables includes the following USEPA provided/derived information: a chemical-specific slope factor (cancer potency factor) for exposures via the oral and inhalation pathway, USEPA's weight-of-evidence cancer classification, and the scientific source of the information.

4.4.3.3 Lead Toxicity

According to USEPA, lead is classified as a B2-probable human carcinogen. However, there is no USEPA value for use as a slope factor in quantifying cancer risks. In the absence of any

USEPA-published toxicity values for lead, it is currently not possible to perform a quantitative risk estimate for lead exposures using standard USEPA methodology. The current USEPA guidance sets forth an interim soil cleanup level for total lead at 400 ppm (USEPA 1998a) which is considered “protective for direct contact at residential settings.” According to USEPA, this guidance adopts the recommendation in the 1985 Centers for Disease Control (CDC) statement on childhood lead poisoning and is to be followed when the current or future land use is residential. The recommendation states that, “...lead in soil and dust appears to be responsible for blood levels in children increasing above background levels when the concentration in soil or dust exceeds 500 to 1,000 ppm.”

Infants and young children are the most vulnerable populations exposed to lead and were the focus of USEPA’s risk assessment efforts. The relatively high vulnerability of infants and children results from a combination of factors: (1) an apparent intrinsic sensitivity of developing organs to lead; (2) behavioral characteristics that increase contact with lead from soil and dust (e.g., mouthing behavior); (3) various physiologic factors resulting in a greater deposition of airborne lead in the respiratory tract and greater adsorption efficiency from the gastrointestinal tract in children than in adults; and (4) transplacental transfer of lead that establishes a lead burden in the fetus, thus increasing the risk associated with additional exposure during infancy and childhood.

For resident children, the risks associated with lead were estimated using the USEPA Integrated Exposure Uptake Biokinetic (IEUBK) Model. The IEUBK Model is used to estimate blood lead concentrations resulting from exposure to environmental sources. The IEUBK Model is a three-stage method for estimating total blood lead levels. First, the intake of lead from each source is assessed. Second, the uptake of lead from each source is determined. Finally, the relationship between the uptake of lead and blood lead concentration is applied using the “Integrated Metabolic Model for Humans of All Ages” (USEPA 1997f). The IEUBK model incorporates total uptake of lead derived from all exposures and the distribution of lead to the four body compartments (blood, bone, liver, and kidney) in which 95 percent of the lead is found. The model was developed based on the distribution and equilibrium of stable lead and a naturally occurring radioactive lead isotope in the bodies of infant and child baboons, and in humans during continuous lead exposure. Model validation has been performed by using data collected from lead smelter sites, by using experimental data on blood lead concentrations in infants, and by studies of lead accumulation in bones under controlled conditions in adults (USEPA 1997f). The model predicts a linear increase in blood lead with increasing lead uptake. However, above a blood lead concentration of 30 μg/dL, the relationship is not linear (USEPA 1997f). Therefore, this model may only be applied for moderately low lead uptakes. USEPA has created a software program of the IEUBK Model, for predicting blood lead levels in children ages 0-84 months, for use on a personal computer. The most current software is LEAD version 0.99D (LEAD99D). LEAD99D was used in this assessment for estimating blood lead levels in children at this site. Standard default values used as input parameters for the model are described in USEPA Guidance (USEPA 1997f). The model output is a probability distribution function describing the percentage of children predicted to have blood levels exceeding 10 μg/dL.

In addition to estimating lead risks for children, the USEPA Technical Review Workgroup for Lead (TRW) has developed a model to predict blood-lead levels in adult workers (USEPA 1996). This model was run to assess potential workers under a commercial setting. Model default parameters were used to predict blood lead impacts for female workers and their potential children at the site.

4.5 Risk Characterization

Risk characterization is the final step of the HHRA process. In this step, the toxicity values were combined with the estimated chemical intakes for the receptor populations to quantitatively estimate both carcinogenic and non-carcinogenic risks. Risks were estimated for the following receptor populations:

- Residents (Adult, Child)
- Adolescent Recreational User
- Construction Workers
- Commercial Workers

The methodologies used to estimate cancer risks and chronic and subchronic hazards for non-carcinogens are described further in the sections below.

4.5.1 Hazard Index for Non-Carcinogenic Effects

The potential human health hazard associated with exposures to non-carcinogenic COPCs at Fort Totten Coast Guard Station were estimated by comparing the ADI with the RfD, as per USEPA Guidance (USEPA 1989). A hazard quotient (HQ) was derived for each COPC, as shown in the equation below:

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

where:

HQ	=	Hazard Quotient; ratio of average daily intake level to acceptable daily intake level (unitless)
ADI	=	Estimated average daily dose (mg/kg-day)
RfD	=	Reference dose (mg/kg-day)

If the average daily dose exceeds the RfD, the HQ will exceed a ratio of one (1.0) and there may be concern that potential adverse systemic health effects will be observed in the exposed populations. If the ADI does not exceed the RfD, the HQ will not exceed 1.0 and there will be no concern that potential adverse systemic health effects will be observed in the exposed populations. However, if the sum of several HQs exceeds 1.0, and the COPCs affect the same target organ, there may be concern that potential adverse systemic health effects will be observed in the exposed populations. In general, the greater the value of the HQ above 1.0, the greater the level of concern. However, the HQ does not represent a statistical probability that an adverse health effect will occur.

For consideration of exposures to more than one chemical causing systemic toxicity via several different pathways, the individual HQs are summed to provide an overall hazard index (HI). If the HI is less than 1.0, then no adverse health effects are likely to be associated with exposures at the site. However, if the total HI is greater than 1.0, separate endpoint-specific HIs may be calculated based on toxic endpoint of concern or target organ (e.g., HQs for neurotoxins are summed separately from HQs for renal toxins). Only if an endpoint-specific HI is greater than 1.0 is there reason for concern about potential health effects for that endpoint.

4.5.2 Cancer Risks

Carcinogenic risk was estimated as the incremental probability of an individual developing cancer over a lifetime as a result of exposure to a potential carcinogen at the site. The numerical estimate of excess lifetime cancer risk was calculated by multiplying the CADI by the risk per unit dose (the slope factor), as shown in the following equation:

$$Risk = CADI \times SF$$

where:

Risk	= The unitless probability of an exposed individual developing cancer
CADI	= Lifetime cancer average daily dose (mg/kg-day)
SF	= Cancer slope factor $(mg/kg\text{-}day)^{-1}$

Because the slope factor is the statistical 95th percent upper-bound confidence limit on the dose-response slope, this method provides a conservative, upper-bound estimate of risk.

It should be noted that the interpretation of the significance of the cancer risk estimate is based on the appropriate public policy. USEPA in the NCP (40 CFR Part 300) (USEPA 1990) states that:

> *"...For known or suspected carcinogens, acceptable exposure levels are generally concentration levels that represent an excess upper bound lifetime cancer risk to an individual of between 10^{-4} and 10^{-6}."*

For the purposes of this risk assessment, the USEPA definition of acceptable carcinogenic risk, the 10^{-4} to 10^{-6} range, will be applied.

4.5.3 Lead Hazard Characterization

The NYDOH stated in comments on the draft RI report that the USEPA's guidance on lead concentrations in soil (Federal Register, Vol. 66, No. 4, 5 January 2001), not the IEUBK model presented in the risk assessment, will be used to evaluate the need for remedial (or other) measures to address potential exposures (see Appendix J). The IEUBK model is part of the basis upon which the USEPA based the *Final Rule for the Identification of Dangerous Levels of Lead*. In Section III-C-2 (page 1216) of the guidance the Agency states "To support the development of the dust and soil hazard standards in this rule, [USEPA] required tools to relate lead in the environment to blood-lead concentration. … The mechanistic model is the Agency's Integrated Environmental Uptake and Biokinetic (IEUBK) model." This is the model used in this risk assessment to assess risks from lead in soil at the upland area. Consequently, there is no difference between the results obtained with the model by USACE and the soil concentrations used in the Final Rule that will be used by NYSDOH.

4.6 Risk Results

The results of the risk assessment for each area are discussed below. Non-cancer hazards are summed across all pathways and all COPC to calculate the cumulative HI:

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

In the event that cumulative HIs exceed 1.0, individual HIs for target organs will be evaluated.

Risk discussion for cancer risks will focus on COPCs that result in cancer risks greater than 1×10^{-6}, which is the lower end of USEPA's acceptable risk range of 10^{-6} to 10^{-4}.

4.6.1 Total Soil, Fill Area

Risk calculations resulting from exposures to total soil in the Fill Area for all receptors and pathways are shown in Appendix A, Tables A-1 through A-17. Risks for receptors of concern exposed to soil are summarized below.

Residential Adults and Children

Non-cancer hazards to future residential adults and children in the Fill Area of the Fort Totten Coast Guard Station are summarized in Tables 4-25 and 4-26, respectively. Detailed calculations shown in Appendix A, Tables A-1 through A-6 for ingestion, dermal, and inhalation of soil particles, and the calculations from consumption of homegrown produce are shown in Tables A-16 and A-17. The total cumulative non-cancer HI for total soil exposure to residential adults is 0.3 (Table 4-25), which is below the hazard target of 1.0. Consequently, non-cancer hazards to adults from exposure to total soil are acceptable. The total cumulative non-cancer HI for residential children is 1.3 (Table 4-26), which is greater than the USEPA hazard target of 1.0. However, examination of Table 4-26 shows that arsenic (21 percent), manganese (21 percent), and mercury (9 percent) represent the greatest contribution to the overall non-cancer risk of 1.3. Table 4-27 shows a target organ assessment for these major contributors, revealing that all three are risk to different target organs. Consequently, once target organs are considered, non-cancer hazards for residential children exposed to total soil in the Fill Area of Fort Totten Coast Guard Station are acceptable.

Cancer risks from exposure to total soil for residential adults and children are shown in Table 4-28. Because cancer risks are averaged over the entire lifetime (assumed to be 70 years), cancer risks for both adults and children were summed together in Table 4-28. The total cancer risk was 3×10^{-5}, within USEPA's acceptable risk range of 10^{-6} to 10^{-4}.

Adolescent Recreational User

Risks for the current and future adolescent recreational user (age 6-15) were quantified as a conservative estimate of risks to all recreational users. Risk calculations for incidental soil ingestion, dermal contact with soil, and inhalation of soil particulate matter are shown in Appendix A, Tables A-7, A-8, and A-9, respectively.

Table 4-29 is a summary of non-cancer hazards for the adolescent recreational user in the Fill Area of Fort Totten Coast Guard Station. The cumulative non-cancer hazard for this receptor is 0.18, which is below USEPA's hazard target of 1.0. Consequently, the adolescent recreational user has acceptable non-cancer hazard from total soil in the Fill Area. Cancer risks for the adolescent recreational user are shown in Table 4-30. The total cumulative cancer risk for this receptor is 4×10^{-6}, within USEPA's acceptable cancer risk range. Arsenic was the only COPC with risks exceeding 10^{-6} (2×10^{-6}). As discussed above, these risks are representative of those expected to be found in an urban location in New York, and do not appear related to site uses.

Risks for adult recreational users are also expected to be acceptable. Risks for the adolescent recreational users represent a conservative upper-limit of both adults and children due to the exposure assumptions inherent in the risk assessment. Because acceptable risks have been found for adolescent recreational users, the same is also true for adult recreational users.

Commercial Worker

Risks for the future adult commercial worker were quantified. Risk calculations for incidental soil ingestion, dermal contact with soil, and inhalation of soil particulate matter are shown in Appendix A, Tables A-10, A-11, and A-12 respectively.

Table 4-31 is a summary of non-cancer hazards for the future commercial worker in the Fill Area of Fort Totten Coast Guard Station. The cumulative non-cancer hazard for this receptor is 0.05; consequently, the future commercial worker has acceptable non-cancer hazard from total soil in the Fill Area. Cancer risks for the commercial worker are shown in Table 4-32. The total cumulative cancer risk for this receptor is 3×10^{-6}, within USEPA's acceptable cancer risk range. Arsenic (2×10^{-6}) was the only COPC in Fill Area total soil showing cancer risks exceeding 1×10^{-6}. As discussed above with respect to residential adults and children, these risks are representative of those expected to be found in an urban location in New York (USEPA 2000b), and do not appear related to site uses.

Construction Worker

Risks for the future adult construction workers were quantified. Risk calculations for incidental soil ingestion, dermal contact with soil, and inhalation of soil particulate matter are shown in Appendix A, Tables A-13, A-14, and A-15, respectively.

Table 4-33 is a summary of non-cancer hazards for the future construction worker in the Fill Area of Fort Totten Coast Guard Station. The cumulative non-cancer hazard for this receptor is 0.2, which is below USEPA's target non-cancer hazard of 1.0. Consequently, the future construction worker has acceptable non-cancer hazard from total soil in the Fill Area. Cancer risks for the commercial worker are shown in Table 4-34. The total cumulative cancer risk for this receptor is 6×10^{-7}, below USEPA's acceptable cancer risk range. Consequently, cancer risks to construction workers from exposure to total soil at the Fill Area are acceptable.

Lead Risk Results

Appendix D contains the model output, both for the residential child IEUBK model (Table D-1) and the adult lead risks (Table D-2). The average lead concentration found in the Fill Area was 417 mg/kg (Table 4-5). Based on the IEUBK model outputs, 93 percent of residential children exposed to lead in total soil are expected to have blood lead levels above the cutoff of 95 percent (Table D-1). Consequently, future resident children are at risk from lead in total soils in the Fill Area. Results of the adult model show that the highest RME blood lead level for developing fetus of adults was found for the construction worker at 6.1 μg/dL, below the cutoff of 10 μg/dL. Consequently, risks to adult receptors from exposure to lead in total soil at the Fill Area are acceptable. Alternatively, residential child exposure to lead resulted in unacceptable risk.

4.6.2 Total Soil, Other Area

Risk calculations for all receptors and pathways are shown in Appendix B, Tables B-1 through B-17. Risks for receptors of concern exposed to soil are summarized below.

Residential Adults and Children

Non-cancer hazards to future residential adults and children are summarized in Tables 4-35 and 4-36, respectively, and the detailed calculations shown in Appendix B, Tables B-1 through B-6, for ingestion, dermal, and inhalation of soil particles, and the calculations from consumption of homegrown produce are shown in Tables B-16 and B-17. The total cumulative non-cancer HI for total soil exposure to residential adults is 0.2 (Table 4-35), below USEPA's hazard target of 1.0. Consequently, non-cancer hazards to adults from exposure to total soil are acceptable. The total cumulative non-cancer HI for residential children is 1.3, greater than USEPA's hazard target of 1.0. As in the fill area, arsenic, manganese, and mercury account for the majority of non-cancer hazards (Table 4-36), and as shown in Table 4-37, target organs for each of these contributors are different. Therefore, with the consideration of target organs, non-cancer hazards for residential children exposed to total soil in the Other Area of Fort Totten Coast Guard Station are acceptable.

Cancer risks from exposure to total soil for residential adults and children are shown in Table 4-38. Because cancer risks are averaged over the entire lifetime (assumed to be 70 years), cancer risks for both adults and children were summed together in Table 4-38. The total cancer risk was 4×10^{-5}, within USEPA's acceptable risk range of 10^{-6} to 10^{-4}.

Adolescent Recreational User

Risks for the current and future adolescent recreational user (age 6-15) were quantified as a conservative estimate of risks to all recreational users. Risk calculations for incidental soil ingestion, dermal contact with soil, and inhalation of soil particulate matter are shown in Appendix B, Tables B-7, B-8, and B-9, respectively.

Table 4-39 is a summary of non-cancer hazards for the adolescent recreational user in the Other

Area of Fort Totten Coast Guard Station. The cumulative non-cancer hazard for this receptor is 0.16, which does not exceed USEPA's hazard target of 1.0. Consequently, the adolescent recreational user has acceptable non-cancer hazard from total soil in the Other Area. Cancer risks for the adolescent recreational user are shown in Table 4-40. The total cumulative cancer risk for this receptor is 5×10-6, within USEPA's acceptable cancer risk range. Arsenic and benzo(a)pyrene were the only COPCs with risk exceeding 10-6.

Commercial Worker

Risks for the future adult commercial worker were quantified. Risk calculations for incidental soil ingestion, dermal contact with soil, and inhalation of soil particulate matter are shown in Appendix B, Tables B-10, B-11, and B-12, respectively.

Table 4-41 is a summary of non-cancer hazards for the future commercial worker in the Other Area of Fort Totten Coast Guard Station. The cumulative non-cancer hazard for this receptor is 0.04; consequently, the future commercial worker has acceptable non-cancer hazard from total soil in the Other Area. Cancer risks for the commercial worker are shown in Table 4-42. The total cumulative cancer risk for this receptor is 4×10-6, within USEPA's acceptable cancer risk range. Arsenic and benzo(a)pyrene risks exceeded 10-6.

Construction Worker

Risks for the future adult construction workers were quantified. Risk calculations for incidental soil ingestion, dermal contact with soil, and inhalation of soil particulate matter are shown in Appendix B, Tables B-13, B-14, and B-15, respectively.

Table 4-43 is a summary of non-cancer hazards for the future construction worker in the Other Area of Fort Totten Coast Guard Station. The cumulative non-cancer hazard for this receptor is 0.2, which does not exceed USEPA's hazard target of 1.0. Consequently, the future construction worker has acceptable non-cancer hazard from total soil in the Other Area. Cancer risks for the commercial worker are shown in Table 4-44. The total cumulative cancer risk for this receptor is

7×10-7, below USEPA's acceptable cancer risk range. Consequently, cancer risks to construction workers from exposure to total soil at the Other Area are acceptable.

Lead Risks

The lead models used to assess risk to receptors at the Fort Totten Coast Guard Station were described in Section 4.6.2. Using these same models, and the average lead concentration in the Other Area of 179 mg/kg (Table 4-6), risks were calculated for children and adults and are shown in Appendix D, Tables D-3 and D-4, respectively. The geometric mean blood lead level for children was 3.8 μg/dL, and the highest adult blood lead level was 3.9 μg/dL for the construction worker. Because these values are well below the risk cutoff of 10 μg/dL, lead risks to all receptors are acceptable at the Fort Totten Coast Guard Station.

4.6.3 Groundwater Risk Assessment for Residential Adults and Children

Risk calculations for residential adults and children for exposure to groundwater by ingestion and dermal contact (adults and children) and inhalation of volatiles (adults) while showering were calculated and are shown in Appendix C, Tables C-1 through C-5. Specific data input and showering model calculations are shown in Appendix E, Tables E-1 through E-3.

Non-cancer hazards from exposure to groundwater for future residential adults and children are summarized in Tables 4-45 and 4-46, respectively, and the detailed calculations shown in Appendix C, Tables C-1 through C-5 for each pathway (ingestion, dermal, and inhalation of volatiles). The total cumulative non-cancer HI for groundwater exposure to residential adults is 6.0 (Table 4-45). The only COPC with an HI exceeding 1.0 was chloroform (HI = 4.6). These hazards are nearly all from the showering scenario from a high concentration of chloroform found in MW2 GW-01-01 in 1998 (23 μg/L). When this same monitoring well was resampled in 2000, the concentration was below the detection limit of 5 μg/L chloroform, and in Spring 2002 a concentration of 5.7 μg/L was measured in this well. Resulting hazards from these later concentrations would be less than 1.0. Non-cancer hazards to adults from exposure to groundwater exceed the hazard target and may represent unacceptable hazards if future use includes residential use of groundwater. The total cumulative non-cancer HI for residential children is 3.6 (Table 4-46). Chromium had an HI of 1.4, as a result of chromium measured in MW1-GW01-01 in Spring 2002 at 51 µg/L. Nickel contributed 18 percent of the non-cancer HI. Non-cancer hazards to children from exposure to groundwater exceed the hazard target and may represent unacceptable hazards if future use includes residential use of groundwater.

Cancer risks from exposure to groundwater for residential adults and children are shown in Table 4-47. Because cancer risks are averaged over the entire lifetime (assumed to be 70 years), cancer risks for both adults and children were summed together in Table 4-47. The total cancer risk was 8×10^{-4}, above USEPA's acceptable risk range of 10^{-6} to 10^{-4}. Benzo(a)pyrene, detected once in MW4-01-01 during Spring 2002 at 10 µg/L accounted for 40 percent of this cancer risk (3×10^{-4}), followed by dibenz(a,h)anthracene representing 29 percent of this risk with 2.2×10^{-4}. This level of risk is the result of an exposure concentration of 2 µg/L (Table 4-7), which was measured in the same well as that for benzo(a)pyrene in 2002. Similarly, this same well had the highest dieldrin concentration, which accounted for a cancer risk of 1×10^{-4}. It is not expected that future use of the site groundwater will include residential use.

4.7 Uncertainty Assessment

There are numerous uncertainties involved in the human health risk assessment process. These are discussed briefly in the following sections.

4.7.1 Sampling and Analysis Uncertainties

The sampling plan can have a significant impact on the results obtained in calculating human health risks at a site. To the extent that samples are taken in areas that are expected to be contaminated (biased sampling), the EPC used in calculating risk exposures and risks is likely to overestimate the actual concentration encountered at the site from random exposure across the site. This sampling bias will generally result in an overestimate of exposures and risks at a site.

4.7.2 Uncertainties Analysis of Exposure Assessment

An analysis of uncertainties is an important aspect of the exposure assessment. It provides the risk assessor and reviewer with information relevant to the individual uncertainties associated with exposure factor assumptions and their potential impact on the final assessment.

A significant uncertainty exists with the basic approach used in arriving at EPCs for the COPCs in soil and groundwater. Uncertainty results from the use of one-half detection limit for all non-detects. An objective of the guidance is to include some quantitative value for COPCs when analytical data indicate that those COPCs were not detected, so that an estimated potential intake and resultant potential risk can be calculated (USEPA 1989). However, this approach generally overestimates the exposure value, and results in overestimates of intakes and subsequent risks, particularly for COPCs with low frequencies of detection.

USEPA has derived a specific process for the derivation of the exposure factors used in this risk assessment (USEPA 1997d). This process involves reviewing the scientific literature supporting these exposure factors, rating the appropriateness of the literature for the derivation, and the selection of upper-limits for their derivation and application. The derivation of residential children and adult incidental soil consumption of 200 and 100 mg/day serves as an example of the conservative nature of these exposure factors. Chapter 4 in USEPA (1997d) documents approximately a dozen scientific studies on the incidental ingestion of soil by children. Various uncertainties have been discussed by USEPA, such as incomplete sample collection (both input and output), the limited length of time that these studies were conducted, the uncertainty associated with the absorption of tracer elements used to derive these values, and the uncertainty associated with heterogeneous soil samples (USEPA 1997d). In particular, it should be noted that all but one of the studies used to derive the 200 mg/day child consumption rate were conducted in summer, when exposure is maximized. The studies are grouped, and typically the upper estimate of the mean chosen as the Reasonable Maximum Exposure (RME) exposure value. The consumption rate for adults (100 mg/day) was based on limited studies (only three), with limited support. Consequently, there is a high degree of uncertainty associated with 100 mg/day Other exposure values shown in Tables 6-10 through 6-19 were selected using similar conservative estimates. A consumption of homegrown produce exposure scenario has been incorporated into residential adults and child exposures. Estimation of the concentrations of chemicals in homegrown produce is problematic, and the plant chemical accumulation factors shown in Table 4-18 all have a high degree of uncertainty.

The use of conservative estimates for exposure including the use of the upper 95th UCLM for the EPC, and the upper limit estimates of each exposure parameter propagates throughout the entire exposure equation:

$$(L)ADI = \frac{C \; x \; IF \; x \; EF \; x \; ED \; x \; RAF}{BW \; x \; AT} \text{ x CF}$$

where:

(L)ADI	=	(Lifetime) Average daily intake (mg/kg-day)
C	=	Concentration in a specific medium (mg/L or mg/kg)
IF	=	Intake factor (mg/day)
EF	=	Exposure frequency (days/year)
ED	=	Exposure duration (years)
RAF	=	Relative absorption factor (unitless)
BW	=	Body weight (kg)
AT	=	Averaging time (days)
CF	=	Conversion Factor (10^{-6} kg/mg)

As an example of this propagation, assume that there is a 5 percent upper limit used for the concentration (C, upper 95th UCLM), and a 10 percent upper limit used for ingestion frequency (IF), exposure frequency (EF), and relative absorption frequency (RAF). The total error then propagates to:

$$Total\ \ Error = 0.05 \times 0.1 \times 0.1 \times 0.1 = 0.00005$$

Thus, while each parameter has selected for the upper 95th or 90th percentile, the resultant (lifetime) average daily intake [(L)ADI] has been calculated at the 99.995th percentile level.

Consequently, the selection of upper limit exposure assumptions for the estimate of RME exposure results in a conservative upper limit dose estimate, adding significantly to the conservative uncertainty of the HHRA.

4.7.3 Uncertainties of Toxicity Assessment

There are numerous uncertainties associated with the toxicity assessment. These are generally due to the unavailability of data to thoroughly calculate the toxicity of COPCs. Two chemicals detected during the Spring 2002 monitoring well samples (p-isopropyltoluene and tert-butyl alcohol) had no toxicity data available. For this reason, risks to these chemicals could not be quantified. It is not known to what degree risks have been underestimated by this data gap. Additional sources of uncertainty are described in more detail in the following sections.

4.7.3.1 Uncertainties Associated With Non-Carcinogenic Effects

Interspecies Extrapolation – The majority of toxicological information comes from experiments with laboratory animals. Experimental animal data were relied on by regulatory agencies to assess the hazards of human chemical exposures. Interspecies differences in chemical absorption, metabolism, excretion, and toxic response are not well understood; therefore, conservative assumptions are applied to animal data when extrapolating to humans. These probably result in an overestimation of toxicity.

Intraspecies Extrapolation – Differences in individual human susceptibilities to the effects of chemical exposures may be caused by such variables as genetic factors (e.g., glucose-6-phosphate dehydrogenase deficiency), lifestyle (e.g., cigarette smoking and alcohol consumption), age, hormonal status (e.g., pregnancy), and disease. To take into account the diversity of human populations and their differing susceptibilities to chemically induced injury or disease, a safety factor is used. USEPA uses a factor between 1 and 10. This uncertainty may lead to overestimates of human health effects at given doses.

Exposure Routes – When experimental data available on one route of administration are different from the actual route of exposure that is of interest, route-to-route extrapolation must be performed before the risk can be assessed. Several criteria must be satisfied before route-to-route extrapolation can be undertaken. The most critical assumption is that a chemical injures the same organ(s) regardless of route, even though the injury can vary in degree. Another assumption is that the behavior of a substance in the body is similar by all routes of contact. This may not be the case when, for example, materials absorbed via the gastrointestinal tract pass through the liver prior to reaching the systemic circulation, whereas by inhalation the same chemical will reach other organs before the liver. However, when data are limited these extrapolations are made, and may result in overestimates of human toxicity.

The USEPA adds uncertainty factors to the RfD whenever there is a source of uncertainty. As shown in Tables 4-20 and 4-21, these uncertainty factors often reduce the RfD by three orders of magnitude, greatly increasing the conservativeness of resultant HHRA risks.

4.7.3.2 Uncertainties Associated With Carcinogenic Effects

Interspecies Extrapolation – The majority of toxicological information for carcinogenic assessments comes from experiments with laboratory animals. There is uncertainty about whether animal carcinogens are also carcinogenic in humans. While many chemical substances are carcinogenic in one or more animal species, only a very small number of chemical substances are known to be human carcinogens. The fact that some chemicals are carcinogenic in some animal species but not in others raises the possibility that not all animal carcinogens are human carcinogens. Regulatory agencies assume that humans are as sensitive to carcinogens as the most sensitive animal species. This policy decision, designed to prevent underestimation of risk, introduces the potential to overestimate carcinogenic risk.

High-Dose to Low-Dose Extrapolation – Typical cancer bioassays provide limited low-dose data on responses in experimental animals for chemicals being assessed for carcinogenic or chronic effects. The usual dose regime involves three dose groups per assay. The first dose group is given the highest dose that can be tolerated, the second is exposed to one-half that dose, and the third group is unexposed (control group). Because this dosing method does not reflect how animals would react to much lower doses of a chemical, a dose-response assessment normally requires extrapolation from high to low doses using mathematical modeling that incorporates to varying degrees information about physiologic processes in the body.

The standard method for modeling high-dose to low-dose effects involves assumptions of extrapolation. Two models are utilized, the default no-threshold model that assumes that the response is linear to the origin (i.e., zero dose, zero response), and a threshold model that assumes that there is some dose above zero, below which there are no adverse effects. It has been found that toxicity data from animal bioassays often fit both no-threshold and threshold models equally well, and it is not possible to determine their validity based on goodness of fit (USEPA 1996b). The dose-response curves derived from these different models diverge substantially in the dose range of interest, with the default no-threshold model yielding much lower toxicity values. Consequently, the model extrapolation process can contribute conservative uncertainty to the chosen toxicity value.

Polycyclic Aromatic Hydrocarbon (PAH) Cancer Uncertainty – Cancer slope factors (SF) have only been determined for the most carcinogenic of those chemical types, benzo(a)pyrene. Due to the absence of chemical-specific SFs for the other carcinogenic PAH, USEPA has utilized Toxic Equivalency Factors (TEFs) to extrapolate SFs for these PAH relative to the benzo(a)pyrene slope factor. There is a significant amount of uncertainty associated with the use of TEFs, which are fixed multiples of the benzo(a)pyrene SF (e.g., 1.0, 0.1, 0.01). There are few scientific justifications for the use of these TEFs, and the use of them is primarily convenience and ease of application.

4.7.4 Uncertainties in Risk Characterization

Uncertainties in the risk characterization can stem from the inherent uncertainties in the data evaluation; the exposure assessment process, including any modeling of exposure point concentrations in secondary media from primary media; and the toxicity assessment process. The individual uncertainties in these respective processes were addressed in previous sections.

4.8 Risk Assessment Summary and Conclusions

A supplemental upland HHRA for Fort Totten Coast Guard Station was conducted to assess potential non-carcinogenic effects and cancer risks from current and future site exposure. Risks to total soils were conducted for two areas, the Fill Area and all Other Areas of the station. Current and future adolescent recreational users, future residential adults and children, future commercial workers, and future construction workers were characterized for risk from ingestion, dermal contact, inhalation of dust, and ingestion of homegrown produce by residents.

Potential risks from consumption of and dermal contact with groundwater was characterized for future residential adults and children. In addition, the risks of volatile chemical inhalation while showering by future residential adults were also quantified. As discussed in the exposure assumptions, it is assumed that inhalation of volatiles by children while bathing does not represent a significant route of exposure; therefore, this was not quantified.

Risks for each area where quantitative HHRA calculations were performed are summarized below.

Fill Area

Non-cancer hazards to future residential adults was less than 1.0, while non-cancer hazards to future residential children were 1.3. These risks were driven by arsenic, manganese, and mercury, which have different target organs. Consequently, future residential children have acceptable non-cancer risks once target organs are accounted. Cancer risks for residential adults and children exceeded 1×10^{-6} (3×10^{-5}) but were within USEPA's acceptable risk range of 10^{-6} to 10^{-4}.

Non-cancer hazards for the current/future adolescent recreational user were less than 1.0. Cancer risk was 4×10^{-6}, within the acceptable cancer risk range. As with residents, arsenic and benzo(a)pyrene accounted for most of this cancer risk. Non-cancer hazards for the future commercial worker were acceptable with a cumulative HI of 0.05. As with the recreational user, cancer risks were slightly above 1×10^{-6}, with arsenic responsible for the majority of this risk. Finally, both non-cancer and cancer risks for the future construction worker were acceptable, with cumulative HI and cancer risks of 0.2 and 6×10^{-7}.

Lead risks were addressed using USEPA's Adult Lead Model for commercial and construction workers were found to be acceptable for adults. The IEUBK showed slightly elevated lead risks for residential children with 6% of potentially affected children having a blood lead level greater than 10 ug/dL, while the standard is 5%. Exposures to lead in soil under a non-residential land use scenario should be acceptable.

Other Area

Concentrations of chemicals and associated risks from these chemicals in the Other Area were very similar to those found in the Fill Area.

As at the Fill Area, the non-cancer hazard index for future residential adults in the Other Area was less than 1.0. The non-cancer hazard index for future residential children exceeded 1.0 at 1.3, with the same risk drivers as found in the Fill Area. Because the non-cancer risk drivers (arsenic, manganese, and mercury) impact different target organs, risks to future residential children from exposure to soil are acceptable. Cancer risks for residential adults and children exceeded 1×10^{-6} (4×10^{-5}) but were within USEPA's acceptable risk range of 10^{-6} to 10^{-4}.

Non-cancer hazards for the current/future adolescent recreational user were acceptable (cumulative HI of 0.16). Cancer risks were 5×10^{-6}, within the acceptable cancer risk range. Non-cancer hazards for the future commercial worker were acceptable with a cumulative HI of 0.04. As with the recreational user, cancer risks were 4×10^{-6}. Finally, both non-cancer and cancer risks for the future construction worker were acceptable with cumulative HI and cancer risks of 0.2 and 7×10^{-7}.

Lead risks were found acceptable for both residential children and adult workers.

Groundwater

Risk calculations were performed for residential adults and children exposed to groundwater. Non-cancer hazards for adults exceeded 1.0 at 6.0. The majority of non-cancer hazard to the adult resident was the result of the inhalation of chloroform (HQ = 4.6). The exposure concentration (23 μg/L) was the maximum of this COPC; however, the exposure-point concentration is less than the MCL for total trihalomethanes of 100 μg/L. Because inhalation risks were not quantified for children, non-cancer hazard for residential children was smaller than that for adults, but still exceeded 1.0 (cumulative HI = 3.6). These risks were from a number of metals, the largest of which was chromium with a HI of 1.4.

Cancer risks from the consumption, dermal contact with, and inhalation of volatiles while showering were above the acceptable cancer risk range (8×10^{-4}). Risks from benzo(a)pyrene and dibenz(a,h)anthracene contributed the majority of these risks at 3×10^{-4} and 2×10^{-4} respectively.

The results of the groundwater future residential adult and child risk assessment indicate that groundwater may not be appropriate for use as a potable water source.

5.0 ECOLOGICAL RISK SCREENING FOR UPLAND AREA

This chapter presents the purpose, rationale, and methods used for the evaluation of ecological risks for the upland areas at the Fort Totten FUDS. A Step 1 and 2 Screening Level Ecological Risk Assessment (SLERA) is a preliminary, initial screening process designed to estimate the likelihood of ecological risk, and to provide a basis for the design of a more thorough assessment, if necessary (USEPA 1997c).

The draft RI report (USACE, 2002), which contained an SLERA for the whole upland area, was reviewed by the State of New York, Department of Environmental Conservation, Division of Fish and Wildlife, the public, and the USACE project manager (i.e., the risk managers for the site). No further sampling activities were required for some portions of the upland area. This portion of the report presents an updated SLERA only for those areas at which supplemental soil data were collected in 2004.

5.1 Objectives

This SLERA follows the same EPA guidance and approach as described in the draft RI report (USACE, 2002). Although the Fort Totten FUDS is not a Superfund site, the EPA guidance provides an accepted framework for ecological risk assessment under any regulatory purview. The overall objectives of the ecological risk Step 1 and 2 screening approach are to characterize the ecological habitat, identify the ecological receptors of concern (ROCs) and constituents of potential concern (COPCs), and to assess the potential for risks to the environment.

The screening level assessment comprises the first two steps of the EPA's eight-step process for ecological risk assessment. The screening level process, as applied to the site, consists of two steps:

1. Problem Formulation and Ecological Effects Evaluation
2. Exposure Estimate and Risk Calculation

The screening level assessment approach corresponds to Steps 1 and 2 in Figure 5-1.

5.2 Problem Formulation and Ecological Effects Evaluation

Problem formulation represents the scoping stage of the ecological risk assessment. Existing information is examined, the site visited and receptors of concern identified, a conceptual model for the site developed to identify potential exposure pathways, and preliminary assessment and measurement endpoints identified. Ultimately, the problem formulation generates one or more questions, speculations, or hypotheses regarding current or future man-induced changes to the environment. These questions are answered or hypotheses tested by collecting information during the analysis phase. The ecological significance of the results is evaluated during risk characterization.

Problem Formulation and Ecological Effects Evaluation consists of the following elements:

- Environmental Setting of the Site
- Identification of Receptors of Concern
- Development of a Conceptual Site Model
- Assignment of Assessment and Measurement Endpoints

Each of these elements is discussed below.

5.2.1 Environmental Setting of the Upland Area

The environmental characteristics of the upland portion of the Ft. Totten FUDS were described in Chapter 2 of the draft RI report (USACE, 2002). The upland land use is best described as an urban campus. The entire upland area is open field habitat with some roads, a parking area, and a ball field. The immediate surrounding land use is similar to the on-site land use. The future use of the site is projected to be recreational as managed by the New York City Department of

Parks and Recreation. Wildlife habitat is limited and use is expected to be transitory and limited to wildlife moving between nearby wetlands and urban residential areas.

5.2.2 Selection of Receptors of Concern

Ecological ROCs are species or guilds of species that are important to the ecology of the study area and that may be susceptible to chemical constituents detected at the site. ROC examples could include an area of riparian wetland, a particular bird species, a benthic community, or a fish. Selection of ROCs is systematic, representative, and ecologically based to ensure that assessment endpoints are addressed adequately. Criteria used to identify ecological ROCs include the following:

- Presence – known or expected to occur onsite
- Susceptibility – exposure pathway is likely complete and of sufficient duration/magnitude
- Representative – of the food web and/or guild
- Data Availability – sufficient and appropriate type of toxicity and exposure information
- Societal Importance – species merits public attention

In some instances, particularly during a screening ERA, the selected ROCs represent an ecological guild (a group of species using similar resources such as food or location in a similar manner).

Ecological ROCs can be classified into three broad categories: (1) ecologically important, (2) recreationally or commercially important, and (3) threatened and endangered species. Ecologically important ROCs substantially contribute to the structure (numbers and biomass) and function (energy flow and nutrient cycling) of the site's ecosystem. This may include primary producers, and primary, secondary, and tertiary consumers, as well as their respective food base. Primary producers are represented by plants, which take energy from sunlight and nutrients from soil pore water. Primary consumers represent the first link of a food web and are represented by soil invertebrates and omnivores such as the white-footed mouse. Secondary consumers consume the primary consumers, and are eaten by tertiary consumers. A bird such as the American crow is an example of a secondary consumer.

For this screening level ERA, four ROCs were chosen for evaluation as described below.

Terrestrial Plants—Plant communities are important to the structure and function of terrestrial ecosystems and are directly exposed to soils on a site. Plants represent the base of food webs, and thus are critical receptors of concern.

Terrestrial Invertebrates (Earthworms)—Soil invertebrates are very important to the fertility of soils because of their role in the aeration and turnover of surface soils. Earthworms will serve as a guild representative for the other invertebrates (e.g., beetles, termites, grasshoppers, butterflies, spiders, moths, and wasps). Earthworms are in continuous contact with any soil-associated contaminants that may be present.

Terrestrial Avian Species—Birds are an important component of the terrestrial community, and several species have been observed at the Fort Totten site. One of these, the American robin, will represent the avian guild in the ERA. The robin is a thrush, and consumes relatively large amounts of invertebrates and fruit.

Terrestrial Mammalian Species—Small mammals would be expected to utilize the limited habitat available in the Fort Totten upland areas. The white-footed mouse will represent the guild of small mammals. The white-footed mouse, and similar species such as the deer mouse, is omnivorous and eats a variety of seeds and other plant material, as well as small arthropods such as insects.

5.2.3 Ecological Risk Conceptual Site Model (CSM)

The CSM is an end product of Problem Formulation (Figure 5-2). It contains a description of the physical and ecological characteristics of the site, potential exposure scenarios, ROCs, and assessment and measurement endpoints.

A major element in every CSM is a description of the exposure scenarios. This consists of four elements:

- Source of COPC and release mechanism(s)
- Transport medium and mechanism of transfer from primary to subsequent media
- Point (or area) of potential ROC contact with the COPC
- Route of uptake by the ROC (ingestion of soil, sediment, food, and bioconcentration)

Potential sources include past activities associated with the Fort Totten FUDS. Because of different historical uses, and potential relative risks, upland source areas in the SLERA were evaluated separately for the PCB Area, Pesticide Area, Fill Area, and Other Area. The Pesticide Area and PCB Area are addressed in the draft RI report and only the Fill Area and Other Area are addressed in this supplemental report. Surface soil is an exposure medium for terrestrial receptors. There could be movement by constituent infiltration to the subsurface soils and to groundwater; however, there are no direct complete pathways for terrestrial ecological receptors to subsurface soil or groundwater. COPCs sequestered in secondary source material (surface soil) may move via several mechanisms, including incorporation into the food web. Terrestrial receptors may directly contact or ingest surface soil at the site. Through the process of trophic transfer, or trophic magnification in the case of bioaccumulative COPCs, biota can serve as vectors for COPC transport up the food chain and expose higher level animals through ingestion.

Exposure routes are based on simple direct contact with surface soil or surface water, or ingestion of soil and plant or prey tissue (Figure 5-2). Exposure pathways and routes include:

- Direct Contact with Surface Soil—This exposure route is important for uptake of COPCs for plants and for soil invertebrates.

- Ingestion of Food (i.e., plants and biota that have taken up constituents from soil)—Herbivores and predators that forage in the terrestrial habitats may ingest plants or animal prey that have bioaccumulated COPCs from surface soils. (In this SLERA, the wet weight concentration of COPC in food items is assumed identical to the dry weight concentration of COPC in surface soil.)

- Incidental Ingestion of Surface Soils—Herbivores and predators that forage in the terrestrial habitats may incidentally ingest some surface soil with their food or during other activities such as grooming. Soil invertebrates ingest surface soil and leaf litter during feeding, but it is difficult to distinguish between uptake as a result of direct contact with surface soils and uptake as a result of ingestion of surface soils because of their intimate association with surface soils. In this SLERA, incidental ingestion by invertebrates is inherently incorporated in the screening benchmarks. Incidental ingestion by vertebrates is incorporated in the very conservative food web models used in this SLERA, as described further below.

5.2.4 Assessment and Measurement Endpoints

USEPA (1998b) guidance stresses the importance of ecologically significant endpoints. The failure to select such an effect for evaluation brings little value to the decision-making process. Several criteria are applicable for assessment endpoint selection (Suter 1993; USEPA 1998b):

- Unambiguous Definition—Assessment endpoints should indicate a subject and a characteristic of the subject (such as American robin reproduction).

- Accessibility to Prediction and Measurement—Assessment endpoints should be reliably predictable from measurements.

- Susceptibility to the Hazardous Agent/Stressor—Susceptibility of an organism (plant or animal) results from the combination of potential for exposure and the sensitivity to the concentrations of contaminants or other stressors of concern.

- Biological Relevance—Biological relevance of impacts to an individual organism is determined by the importance of the impact to higher levels of biological organization such as populations or communities.

- Social Relevance and Policy Goals—Assessment endpoints should be of value to decision-makers and the public. The assessment endpoints should represent an effect that would warrant consideration of site remediation or alteration of project plans. Assessment endpoint selection should also include endpoints that may be mandated legally (e.g., protected species).

The extent to which these items are considered varies from site to site, and it depends on several factors including the level of public involvement, the ecological character of the site, and the lead regulatory agency involved in the assessment.

The selection of assessment endpoints must be based on the fundamental knowledge of the local ecology. Assessment endpoints typically relate to an effect on a population or community. Survival of a specific species of earthworm is an example of a population level assessment endpoint. Community level assessment endpoints could include survival of all soil invertebrates or the primary productivity of vegetation found at a site. Examples of endpoints representing guilds of species are useful in that they convey information beyond the indicator species identified in the endpoint itself.

Based on previous activities at the Fort Totten FUDS site, ecological ROCs may be exposed to COPCs through surface soil exposure. COPCs previously detected in the soil at this site may be ingested via soil and food (i.e., plants and biota that have taken up constituents from soil).

Based on the above observations, the following ecological assessment endpoints are defined:

1. Protection of plant communities to ensure that COPCs in soil do not have unacceptable adverse effects on survival, growth, and reproduction of key plant species, which may result in adverse effects to the community structure such as diversity or biomass.

2. Protection of soil-invertebrate communities to ensure that COPCs in soil do not have unacceptable adverse effects on survival, growth, and reproduction of key soil

invertebrate species, which may result in adverse effects to the community structure such as diversity or biomass.

3. Protection of mammals, represented by the small terrestrial omnivorous white-footed mouse to ensure that ingestion of COPCs in food items and soil/sediment does not have unacceptable adverse impacts on survival, growth, and reproduction.

4. Protection of birds, represented by the omnivorous American robin, to ensure that ingestion of COPCs in food items and soil/sediment does not have unacceptable adverse impacts on survival, growth, and reproduction.

Measurement endpoints are measurable ecological characteristics that are related to the assessment endpoints (USEPA 1998b). Because it is difficult to "measure" assessment endpoints, measurement endpoints were chosen that permit inference regarding the above-described assessment endpoints. Measurement endpoints selected for this risk assessment include:

- Media Chemistry for Surface Soils—The measurement of chemical constituent concentrations in surface soil provides the means, when compared to appropriate background and ecotoxicological-based screening concentrations, for drawing inferences regarding the first measurement endpoint above. Because soil invertebrates and plants are in direct contact with the soil, direct measurement of soil concentrations is an appropriate endpoint.
- Calculated Dietary Doses—Measurement endpoints to address calculated chemical doses in the diet for birds and mammals. The knowledge of specific COPC concentrations in surface soil cannot be used to address this assessment endpoint directly. Rather, these measurements are used in conjunction with food ingestion rate and other factors to calculate the daily intake, or dietary dose of a constituent. These are then compared to toxicological thresholds to address the assessment endpoint, as described in Section 8.6 below. Because this SLERA, the assumption has been made that food concentrations are equivalent to those found in the surface soil. This assumption is conservative for the

majority of compounds; however, for compounds known to bioaccumulate.

5.3 COPC Screen

A risk assessment begins with a list of analytes that include compounds and/or elements known or suspected to have originated from site-related activities. Depending on the area in question at the Fort Totten site, these are metals, PAHs, other SVOCs, and pesticides. Analytes not detected or at non-hazardous concentrations may be candidates for elimination. Analytes known or suspected to have originated from site-related activities remaining after the screening process are COPCs.

The screening process that identifies COPCs must not eliminate analytes that could pose potential ecological risk. In statistical terms, the screening process must minimize the potential for false negatives. This potential is minimized by using conservative assumptions and appropriate screening values during the COPC screening process. If possible, these screening values should be toxicologically based, as discussed below.

On a national basis, USEPA has only a limited number of ecologically based soil screening values. Screening values recommended for soil were taken from the draft USEPA Soil Screening Level documents (USEPA 2000b), Oak Ridge National Laboratory (ORNL) (Efroymson et al. 1997a and 1997b), RIVM (1994, 1995, 2000), and in this order. Some special references were found in the scientific literature for analytes not contained in these sources. Appropriate values from these sources are shown as ecological soil screening values.

The maximum site concentrations in surface soil sample (0-24 inches) were compared to the corresponding ecological screening values. (In cases were a duplicate sample was collected, the higher of the pair was selected.) The comparison was done by dividing the site maximum by the screening value to produce a Hazard Quotient (HQ). The HQ is a unitless ratio that reflects the relationship of the site concentration to the screening value. If the site maximum was less than the screening value (HQ < 1.0), that analyte was eliminated as a COPC. If the site maximum exceeded the screening value (HQ > 1.0), that analyte was retained as a COPC. In the latter case, the HQ reflects the magnitude of exceedance of the screening value by the site concentration. Calcium, iron, magnesium, potassium, and sodium were included in the screening tables, but were not considered as COPCs because of their importance as essential nutrients.

The identification of ecological COPCs in surface soil for the Fill Area and the Other Area is summarized in Tables 5-1 through 5-2, and discussed below.

Fill Area

Metals (e.g., aluminum, chromium, copper) produced HQs greater than 1.0 and were retained as COPCs (Table 5-1). Several organic constituents, including carbazole and dibenzofuran, were retained as COPCs because there were no available screening values.

Other Area

Metals (e.g., aluminum, chromium, cobalt), total PAHs and total phthalates were retained as COPCs because the HQs were greater than 1.0 (Table 5-2). Five organic compounds (e.g., 1,2-dichlorobenzene, carbazole, dibenzofuran) were retained as COPCs because of a lack of screening values.

5.4 Step 1 And 2 Exposure Assessment

Exposure assessment is a key component of risk quantitation, linking contaminants to receptors through complete pathways. Exposure refers to the degree of contact between ecological receptors at a site and the COPCs.

Based on the CSM, terrestrial receptors at the Fort Totten FUDS were assumed to be exposed to COPCs in surface soil either through direct contact, or via dietary food web. In either case, the starting point for the evaluation of terrestrial receptors is the concentration in the surface soil.

The relevant pathway for terrestrial plant and soil invertebrate communities is chronic exposure to surficial soil contaminants that may exhibit a detrimental effect on survival and growth. This exposure assessment was very conservative and was set up such that soil concentrations were compared to the lower of available vegetation or invertebrate screening values. It is assumed that the COPCs are 100 percent bioavailable for uptake by plants and invertebrates. Risk to terrestrial plants and soil invertebrates is based on calculation of an HQ.

Hazard Quotient = Maximum Surface Soil Concentration / Plant/ Invertebrate Screening Value

The relevant pathway for terrestrial mammalian and avian ROCs is chronic exposure to surficial soil contaminants due to dietary uptake. The ROCs occupy different feeding guilds, but have diets that contain potential vectors for site-related soil contaminants.

Consistent with USEPA guidance (USEPA 1997c), COPC concentrations in food organisms were assumed to be at the same concentrations as the soil. This exposure is particularly conservative. It substitutes soil for vegetation, invertebrates, or mammals that organisms would typically ingest as their main food items. In addition, it assumes that all food is on a dry-weight basis, but this food is consumed at a much higher wet-weight basis; consequently, dietary doses (and therefore the resulting hazards) are overestimated.

Dietary exposures for ROCs have been estimated as body-weight-normalized daily doses for comparison to a body-weight-normalized daily dose toxicity reference value (TRV). The daily dose for a given receptor to a given COPC is given by summing the products of feeding rate and food items and multiplying the sum by the total feeding rate and a habitat usage factor (assumed to be 100 percent for this food web). Separate doses are presented for soil and food contributions, and these are summed to produce the total dose for each ROC.

$$Dose_{total} = Dose_{food} + Dose_{soil}$$

where:

$Dose_{total}$ = Total daily dose of COPC received by receptor; mg COPC/kg-bw/day

$Dose_{food}$ = Daily dose of COPC received by receptor; mg COPC/kg-bw/day from food items
$Dose_{soil}$ = Daily dose of COPC received by receptor; mg COPC/kg-bw/day from incidentally ingested soil

The total dose from food is given by:

$$Dose_{food} = F_f \; x \; U \; x \; C_f$$

where:

F_f = Total daily feeding rate in kg food/kg-body weight of ROC/day (wet basis)
U = Habitat usage factor (fraction of habitat range represented by site) for receptor; assumed to be 1.0 for this food web
C_f = Concentration of COPC in food; assumed to be the same concentration as soil (mg chemical/kg food)

The total dose from incidental soil is given by:

$$Dose_{soil} = F_s \; x \; U \; x \; C_s$$

where:

F_s = Total daily incidental soil feeding rate in kg soil/kg-body weight of ROC/day (wet basis)
U = Habitat usage factor (fraction of habitat range represented by site) for receptor; assumed to be 1.0 for this food web
C_s = Concentration of COPC in soil; mg chemical/kg soil (dry basis)

Lastly, the total daily soil feeding rate is given by:

$$F_s = F_f \; x \; F_{xsoil}$$

where:

F_s = Total daily incidental soil feeding rate in kg soil/day (wet basis)

F_f = Total daily feeding rate in kg food/day (wet basis)
F_{xsoil} = Fraction incidental soil ingestion as a proportion of food ingestion rate

Information necessary for calculation includes organism body weight (BW), food ingestion rate (F_f), fraction incidental soil ingestion as a proportion of food ingestion rate (F_{xsoil}), and analyte concentrations of ingested materials. As discussed earlier, vegetation and animal food items were represented by the same concentration as found in soil (dry weight). Information specifically relevant to the ecology of the ROCs (i.e., body weights, food ingestion rates, and incidental soil ingestion rates) was obtained from published sources. The primary source used for these exposure parameters was the Exposure Factors Handbook (USEPA 1993).

5.5 Toxicity Assessment

This section summarizes the screening values and TRVs used in this ecological risk assessment. USEPA (1997c) guidance specifies that a screening ecotoxicity value should be “equivalent to a documented or best conservatively estimated chronic No Observed Adverse Effect Level (NOAEL).”

5.5.1 Soil Invertebrates and Terrestrial Plants

Risks to soil invertebrates and terrestrial plants are assessed relative to soil concentrations, using the screening values employed in the COPC screen. As available, screening values were obtained in order from USEPA (2000b), Oak Ridge values (Efroymson et al. 1997a, 1997b), and the Dutch values (RIVM 1994, 1995, 2000). If both plant and invertebrate screening values were available for a given analyte, the lower of the two was chosen.

5.5.2 TRVs for Terrestrial Food Web Risks

The terrestrial ROCs that were selected include both avian species (American robin) and mammalian species (white-footed mouse). Food web risks for avian and mammalian species are expressed relative to a dose of chemical (mg/kg body weight/day) taken up by the organism from food and soil. Literature-reported wildlife NOAEL TRVs (Sample et al. 1996) were primarily used as TRVs for the terrestrial food-web risks.

As noted in Sample et al. (1996), the current state of avian toxicology indicates that the use of allometric relationships, used to relate the body weight of the toxicity test organism to that of the receptor of concern, are not appropriate. Consequently, toxicity values for avian ROCs taken from Sample et al. (1996) are the same regardless of the receptor of concern, and are equivalent to that found in the test species (pheasant, chickens, and ducks). An allometric conversion was performed to modify the toxicity value from the test species to mammalian ROCs (Sample et al. 1996). This is due to the finding that smaller animals, such as rats and mice that are commonly used as test species in toxicity tests, have higher metabolic rates, and detoxify contaminants faster than larger animals.

Example Food-Web Calculation

An example HQ calculation provided below estimates the potential for risk for the case where the white-footed mouse is exposed to soil in the Fort Totten FUDS Other Area containing the maximum concentration of lead. The maximum concentration of lead reported in surface soil (dry-weight basis) in the Other Area was 793 mg/kg (Table 5-2).

The following equation provides the dose to the receptor from food ingestion:

$$Dose_{food} = F_f \; x \, U \; x \, C_f$$

= (0.1989 kg/kg-bw/day x 1.0 x 793 mg/kg)
= 157.73 mg/kg-bw/day

where:

F_f	=	Total daily feeding rate in kg food/kg bw of ROC/day (wet basis)
U	=	Habitat usage factor (fraction of habitat range represented by site) for receptor; assumed to be 1.0 for this food web
C_f	=	Concentration of COPC in surface soil

The dose from incidental soil ingestion is calculated using:

$$Dose_{soil} = F_s \times U \times C_s$$

= (0.00398 kg/kg-bw-day x 1.0 x 793 mg/kg)

= 3.15 mg/kg-bw/day

where:

Fs = Total daily incidental soil feeding rate in kg soil/kg-bw of ROC/day (wet basis)

U = Habitat usage factor (fraction of habitat range represented by site) for receptor; assumed to be 1.0 for this food web

Cs = Concentration of COPC in soil; mg chemical/kg soil (dry basis)

The final dose is calculated as follows:

$$Dose_{total} = Dose_{soil} + Dose_{food}$$

$Dose_{total} = 3.15 + 157.73$

$Dose_{total} = 160.89$ mg/kg-bw/day

The HQ is calculated from the dose and the TRV as follows:

$$HQ=Dose/TRV$$

HQ=160.89/15.98

HQ=10.08

The TRVs and the final HQs can be found in Tables 5-3 through 5-6.

5.6 Step 2 Risk Characterization

Risk characterization is primarily a process of comparing the results of the exposure assessment with the results of the ecological effects assessment. Available methods (either quantitative or qualitative) seek to answer the following questions:

- Are ecological receptors currently exposed to site contaminants at levels capable of causing harm, or is future exposure likely?
- If adverse ecological effects are observed or predicted, what are the types, extent, and severity of the effects?
- What are the uncertainties associated with the risk assessment?

The risk characterization concludes with a risk description, which (1) includes a summary of the risks and uncertainties, and (2) interprets the ecological significance of the observed or predicted effects. The risk description is a key step in communicating ecological risks to site managers and decision makers. The final statement regarding the potential for risks to ecological receptors considers such factors as the nature and magnitude of the effects, the spatial and temporal distribution of the effects, and the potential for recovery.

5.6.1 Soil Invertebrates and Terrestrial Plants

In this screening level ERA, risks to lower trophic level organisms such as plants and invertebrates are established in the COPC-screening process. As indicated, when both plant and invertebrate screening values were available for a given analyte, the lower of the two was used in the COPC screen. This afforded a conservative assessment of risks to lower trophic levels, as well as establishing COPCs for further evaluation in the Step 2 food-web assessment. The data from screening Tables 5-1 to 5-2 are summarized below to reflect the potential risk to lower trophic-level plants and invertebrates, based on screening HQs greater than 1.0.

	Fill Area	**Other Area**
Aluminum	X	X
Antimony	X	
Chromium	X	X

	Fill Area	Other Area
Cobalt		X
Copper	X	X
Lead	X	X
Manganese	X	
Mercury	X	X
Selenium	X	X
Silver		X
Vanadium	X	X
Zinc	X	X
Total PAH		X
Total phthalates		X

5.6.2 Food-Web Risks to Wildlife: Mammals and Birds

The food-web risk characterizations include several conservative assumptions. It was assumed that prey items have the same dry-weight concentration as the maximum soil concentration of COPCs on the site. In addition, wet-weight consumption quantities were used with dry-weight soil concentrations. COPCs are assumed to be 100 percent bioavailable. That is, all of the COPCs are available for absorption and expression of toxic effects. These assumptions are conservative and contribute to the conservative nature of the risk characterization and to probable overestimation of risk at this stage in the SLERA process.

Risks to terrestrial receptors were grouped based on the absolute value of HQs, or the ratio of exposure to TRVs. Because of the conservative assumptions built into a SLERA, risks are interpreted as follows:

- HQ less than 1.0 indicates no unacceptable risk for the receptor/analyte pair. Given the conservative assumptions used for this food web, the probability of false negatives (the potential of finding acceptable risk when there is unacceptable risk) is very small.

- HQ greater than or equal to 1.0, but less than 10. Because of the exposure assumptions used in this food-web screen, correction of obvious inaccuracies such as ingestion of soil measured in dry weight on a wet-weight basis would be expected to easily decrease exposure by an order of magnitude.

- HQ of greater than 10 but less than 100 indicates potential risk levels based on the exposure models and toxicity data used in the assessment.

- HQ greater than 100 represents a high level of potential risk to ecological receptors from exposure to the COPC based on the exposure models and toxicity data used in the assessment.

For some COPCs, there are no available TRVs. These COPCs cannot be eliminated as of concern, although they cannot be quantified. Such a COPC must be considered as a potential risk through dietary exposure.

The results of the food-web risk calculations are shown in Tables 5-3 through 5-8. A number of HQs exceed 1.0 for both the mouse and robin in the fill and other areas, and the robin in the pesticide area. Based on the HQ-interpretive scheme presented above, the food-web results for each area and ROC are summarized as follows:

	No. of Chemicals in the Fill Area with an HQ within the Specified Range	No. of Chemicals in the Other Area with an HQ within the Specified Range
Mouse		
HQ 1-10	1	0
HQ 10-100	3	4
HQ > 100	1	0
Robin		
HQ 1-10	4	3
HQ 10-100	3	4
HQ > 100	2	1

Several of selected receptors have HQs that are greater than 10 and four HQs exceed 100. The majority of the chemicals had HQs in the middle of the range. In addition, several chemicals had no toxicity data with which to calculate an HQ.

5.7 Uncertainty Assessment

Ecological risk characterization includes analysis of uncertainty (USEPA 1997c). Uncertainty is distinguished from variability, and arises from lack of knowledge about factors associated with the study. In a screening-level assessment such as this one, uncertainty typically stems from two study facets: the sampling plan and the toxicological data. Sources of uncertainty can include the process of selecting COPCs, assumptions made in establishing the CSM, adequacy of ecological characterization of the site, estimates of toxicity to receptors, and selection of model parameters. There are a number of factors that contribute to uncertainty in characterization of ecological risk in the upland portion of the Fort Totten FUDS, as described below.

Environmental media are typically sampled in a non-random fashion. That is, sampling points are chosen to best characterize known or suspected areas of contamination. Peripheral and nearby areas are undersampled, if at all, and thus the average exposure of ecological receptors is biased high. A SLERA uses the maximum measured concentration to estimate risks consistent with guidance, which represents a high bias in exposure to ROCs.

Toxicological data used in the risk characterization represent significant uncertainty. Because there may be no known data on the effects of chemical constituents on specific ROCs, toxicological data for surrogate species are sometimes used, and this adds uncertainty.

Food-item concentrations were overestimated. Plant and animal food items had not been sampled at the site and no bioaccumulation factors were used to estimate the chemical concentrations in food items. The extremely conservative assumption was made that all food (vegetation, soil invertebrates, etc.) was at the same concentration as the dry-weight soil or sediment maximum. Based on a review of published bioaccumulation factors for many of the COPCs identified in this assessment, these assumptions are conservative by factors of 10 to more than 100.

- Food item concentrations were expressed on a dry-weight basis. The food ingestion rates used from the *Wildlife Exposure Factors Handbook* (USEPA 1993) are ingested food on a wet-weight basis. Because dry-weight-basis soil was directly applied as food concentrations for food items, the exposure to receptors that consume plants and animals (robin, mouse) was overestimated. Percent moisture in food items is commonly 50 percent or greater, thus the use of dry-weight food results in an artificial increase of chemical ingestion of at least 100 percent.

- Incidentally ingested soil concentrations are expressed on a dry-weight basis. USEPA (1993) clearly notes that the fraction of incidental soil ingestion should be on a wet-weight basis, and recommends that the wet food ingestion rate be converted to a dry food ingestion rate prior to calculation of dose. In conformance with USEPA (1997c) this was not performed for this screening assessment. Therefore, this assessment overestimates incidental ingestion.

COPCs were assumed to be 100 percent bioavailable. The assumption that COPCs are 100 percent bioavailable is highly unlikely based on soil chemistry. Elements such as lead, manganese, and zinc are common constituents of soil. In the solid soil matrix, most of these elements are not bioavailable, and are thus not taken up into organisms exposed to these soils. The environmental behavior (and thus the bioavailability) of metals in environmental soils is complex and not well understood. The solubility and availability of these metals is dependent on a number of factors including soil Eh (a measure of the oxidation/reduction potential), pH, and availability of ligands (chemical constituents capable of bonding with metal ions) (Bodek et al. 1988b).

The toxicological data that underpin the screening values are inherently uncertain because laboratory data are extrapolated to specific field sites. This uncertainty is to some extent controlled by choosing the lowest available screening values, consistent with USEPA (1997c) guidance to "be consistently conservative in selecting literature values…".

Although the direction of bias of some uncertainties is unknown, the overriding influence of the non-random media sampling and assumptions of 100 percent bioavailability assures that risks are overestimated.

5.8 Summary Of Steps 1 And 2 Ecological Risk Screening

As reflected in HQs greater than the benchmark of 1.0, the Step 1 and 2 ERA screening process identified potential risk from a number of analytes for several receptors over several upland areas of the Fort Totten site. As identified in the Step 1 COPC-screening process, potential risk to lower trophic level organisms (i.e., plants and/or soil invertebrates) were identified for from 1 to 12 analytes, depending on the area of Fort Totten in question. A number of the HQs were less than 10, reflecting minimal risk. However, some were greater than 10, indicating potential risk to lower trophic levels (i.e., aluminum, lead, mercury, and vanadium at both the fill area and other area [Tables 5-1 and 5-2]). Similarly, food-web calculations projected at least potential risk, and in limited cases the HQ was greater than 100 for some receptor/analyte pairs.

Risk assessment guidance (USEPA 1997c) calls for a Scientific Management Decision Point (SMDP) following the SLERA process. The purpose of the SMDP is to generate communication between risk assessor and risk manager to evaluate the results of the screening ERA and generate a decision regarding whether a site does or does not represent unacceptable ecological risk, or whether addition information is needed to support the decision.

The draft RI report, which contained an SLERA for the upland area, was reviewed by the State of New York, Department of Environmental Conservation, Division of Fish and Wildlife, the public, and the USACE project manager (i.e., the risk managers for the site). The SLERA in the draft RI report was updated with the soil data collected in 2004. In light of the potential risks, uncertainties, and conservative nature of the SLERA for the upland areas, the risk managers concluded, in the context of the SMDP process, that further efforts to define the risks to upland ecological receptors was not warranted. This initial conclusion is further supported by the additional soil data collected in 2004. USEPA guidance for ecological risk assessment (1997c) recognizes that because of the very conservative exposure assumptions used in Steps 1 and 2, COPCs may be identified that in fact pose negligible risk. It is the conclusion of this SLERA that refinement of the hazards identified in the SLERA is not warranted because:

- Comparison of site concentrations to natural and anthropogenic background concentrations would likely show similar risks for ecological receptors;

- Overestimation of the exposure concentration through use of the maximum measured concentration;

- Overestimation of the food-web doses on a wet-weight basis rather than the natural moisture content of food items – a more realistic exposure scenario than the dry weight basis used in the Step 2 food web;

- The area use factor was assumed to be 1.0 resulting in the assumption that receptors forage at no other location except on the upland area all year long, even in the case of species that are not present all year long (e.g., robins).

6.0 SUMMARY AND RECOMMENDATIONS

The SRI report presented and discussed the results of the SRI conducted at the Fort Totten Coast Guard Station in Queens, New York during July and August 2004. The USACE performed the SRI to further delineate and evaluate contamination in soil and groundwater within the upland portion of the property and the indoor air in Building 615. This investigation represented the final phase of the environmental investigation conducted under the USACE's Formerly Used Defense Site (FUDS) program. This portion of the environmental investigation was initiated in response to comments from the NYSDEC and NYSDOH (see Appendix J) on the draft RI report (USACE 2002). The state agencies' comments focused on the need to collect additional samples of soil, air, and groundwater at specific locations on the FUDS property.

6.1 Data Collection And Analysis

The upland area has been the subject of several previous investigations. USACE initiated a comprehensive remedial investigation in 1997 to determine the nature and extent of the contamination reported in earlier studies. This RI was conducted in two phases. Phase I was conducted from July of 1997 through August of 1998 and included the collection and analysis of soil, sediment, and groundwater samples. In 1998, a removal action was completed by the USACE inside Building 615 to remove sumps, drains, and discharge pipes that were contaminated with mercury. USACE performed Phase II from November 1999 through August 2000 to obtain more detailed information about the site. The investigation involved the collection of additional sediment, groundwater, surface water and surface soil samples from the same or similar locations as Phase I. In the spring of 2002, the USACE collected an additional round of groundwater samples from the five existing monitoring wells.

During Phases I and II, USACE had collected and analyzed 92 soil samples from 70 different locations in upland areas of the Coast Guard Station. The investigation showed that concentrations of PAHs and metals in some surface soil samples were greater than the screening concentrations used by the NYSDEC. Levels of metals detected in the groundwater were found to be in the range expected to occur naturally in the environment; however, low levels of PAHs were found in the groundwater at MW-4.

The USACE conducted a supplemental investigation in summer 2004 to address data gaps and questions raised by the NYSDEC and NYSDOH regarding the upland portion of the Coast Guardfacility and Building 615. The upland areas are defined as those areas on the FUDS property that are east of the Little Bay shoreline and west of the North Loop road. Figure 1-3 shows the locations of Building 615, the Fill Area and other areas that make up the "upland portion of the facility that were addressed in this SRI.

The Spring 2004 soil sampling program was executed to obtain additional data on levels of SVOCs and metals from the upland areas. Twenty-two soil samples were collected from 11 new soil boring locations at a depth of 12 to 24 inches. The soil sample locations were selected based on the presence of higher concentrations of SVOC levels during the previous sampling and as requested by the NYDEC.

In order to address state concerns regarding SVOCs in the groundwater near MW-4, a replacement monitoring well was installed. The new monitoring well was developed and a groundwater sample was taken and analyzed for SVOCs.

The final element of the investigation focused on ambient indoor air monitoring for mercury inside Building 615. Samples were taken with a real-time monitor and fixed-based samplers at heights of three feet and six feet above each of the two floors.

During collection of indoor air samples, a floor drain inside of Building 615 was located (Figure 2-4). Sediment/sludge samples were collected from within the pipe. In addition, real time and fixed based indoor air sample were also collected at this location.

6.2 Data Evaluation Summary

The data were collected in accordance with the state approved Work Plan. Samples were analyzed in accordance with the approved Quality Assurance Project Plan by New York State certified laboratories. Subsequent to data analysis by the laboratory the data were validated in accordance with state and USEPA requirements by an independent third party data validator (see Appendix F).

Split samples were collected for quality control purposes and duplicate samples were collected for quality assurance purposes. A Data Usability Report was prepared by the third party data validator which found no major data quality concerns. In addition, a Chemical Quality Assurance Report was also prepared (Appendix G). Therefore, the data were deemed acceptable for site characterization and risk assessment.

USAE reviewed potential ARARs and TBC items to determine appropriate screening criteria for the comparison with site data. No ARARs for soil or indoor air were located. The ARAR for groundwater was determined to be provided in the state regulations, 6 NYCRR Part 703.5.
The following potential TBC guidelines were identified for response activities at the upland area:

- NYSDEC Ambient Water Quality Standards and Guidance Values, Division of Water, Technical and operational Guidance Series (TOGS) 1.1.1, (June ’98).

- Determination of Soil Cleanup Objectives and Cleanup Levels, NYDEC, Technical Assistance Guidance Memorandum (TAGM) HWR-94-4046 (revised).

In addition, the NYDEC and NYDOH provided an indoor air screening level of 1 $\mu g/m^3$ for mercury in indoor air. Occupational benchmarks for mercury in indoor air include:

- OSHA PEL – 100 µg/m^3 of air as a ceiling limit.
- NIOSH REL – 50 µg/m^3 as a TWA for up to a 10-hour workday and a 40-hour workweek.
- ACGIH TLV – 25 µg/m^3 as a TWA for a normal 8-hour workday and a 40-hour workweek.

The results of the groundwater sampling and laboratory analysis indicated that low concentrations of SVOCs are present in the groundwater near MW-4R. The groundwater at Ft. Totten is classified "GA", according to NYSDEC regulations. The concentrations of SVOCs in the new groundwater sample from MW-4R do not exceed the State promulgated value for any compound. The groundwater results are presented in Table 3-3.

The results of the surface soil sampling and laboratory analysis indicated that low concentrations of SVOCs and elevated concentrations of metals are present in the surface soil at concentrations greater than the state screening levels. The results are presented in Tables 3-1 and 3-2, for SVOCs and metals, respectively. Concentrations of chemicals and associated risks from these chemicals in the Other Area were very similar to those found in the Fill Area.

The results on the onsite (real time) indoor air monitoring indicated that there were no detectable concentrations of mercury greater than the state screening level of 1 µg/m^3 in the indoor or background air samples. The results on the off-site laboratory sample analysis indicated that there were no detectable concentrations of mercury.

The results of the analysis of the material (sludge) that was collected from within the floor drain pipe in Building 615 indicate that mercury is present. Analysis of the material inside of the drain pipe indicated mercury concentrations of 6.25 to 10.0 mg/kg. The exact discharge point of the floor drain pipe is outside of Building 516; however, the exact location is unknown. A mercury concentration of 0.082 µg/m^3 was reported at this location. Fixed based air samples at 3 ft and 6 ft above the floor at this location were nondetect. Dye testing of the drain pipes does not indicate a connection with Little Bay

6.3 Human Health Risk Assessment Summary

A supplemental HHRA for upland portion of the FUDS was conducted to assess potential non-carcinogenic effects and cancer risks from current and future site exposure.

Hazards and risks from exposure to total soils were conducted for two areas, the Fill Area and Other Area in the upland portion of the FUDS. The groundwater at the FUDS was evaluated as one exposure unit. Hazards and risks were estimated for multiple receptors including current and future adolescent recreational users, future residential adults and children, future commercial workers, and future construction workers. Potential health risk from exposure to chemicals in groundwater was characterized for future residential adults and children.

The potential health risks and hazards for each portion of the upland area are summarized below.

6.3.1 Fill Area

Non-cancer hazards to future residential adults was less than 1.0, while non-cancer hazards to future residential children were 1.3. These risks were driven by arsenic, manganese, and mercury. Cancer risks for residential adults and children exceeded 1×10^{-6} but were within USEPA's acceptable risk range of 10^{-6} to 10^{-4}.

Non-cancer hazards for the current/future adolescent recreational user were less than 1.0. Cancer risk was 4×10^{-6}, within the acceptable cancer risk range. Exposure to arsenic and benzo(a)pyrene in soil accounted for most cancer risk.

Non-cancer hazards for the future commercial worker were acceptable with a cumulative HI of 0.05. As with the recreational user, cancer risks were slightly above 1×10^{-6}, with arsenic responsible for the majority of this risk. Both non-cancer and cancer risks for the future construction worker were acceptable, with cumulative HI and cancer risks of 0.2 and 6×10^{-7}.

The assessment indicated slightly elevated lead risks for residential children with 6% of potentially affected children having a blood lead level of greater than 10 ug/dL, while the standard is 5%. Exposures to lead in soil under a non-residential/non-commercial land use (e.g., recreational) scenario should be acceptable. Hazards from exposure to lead for commercial and construction workers were found to be acceptable for adults.

The HHRA indicates that the Fill Area of the Ft. Totten FUDS presents an unacceptable hazard under a residential reuse scenario due to lead. Therefore, it may be in appropriate for future unrestricted reuse.

6.3.2 Other Area

The non-cancer hazard index for future residential adults was less than 1.0. The non-cancer hazard index for future residential children exceeded 1.0 at 1.3, with the same risk drivers as found in the Fill Area (arsenic, manganese, and mercury). Cancer risks for residential adults and children exceeded 1×10^{-6} (4×10^{-5}) but were within USEPA's acceptable risk range of 10^{-6} to 10^{-4}.

Non-cancer hazards for the current/future adolescent recreational user were acceptable (cumulative HI of 0.16). Cancer risks were 5×10^{-6}, within the acceptable cancer risk range. Non-cancer hazards for the future commercial worker were acceptable with a cumulative HI of 0.04. As with the recreational user, cancer risks were 4×10^{-6}. Finally, both non-cancer and cancer risks for the future construction worker were acceptable with cumulative HI and cancer risks of 0.2 and 7×10^{-7}.

Hazards from exposure to lead were found to be acceptable for residential children and adult commercial and construction workers.

The HHRA indicates that the Other Area of the Ft. Totten FUDS does not present an unacceptable risk under the multiple scenarios evaluated and may be considered for unrestricted use.

6.3.3 Groundwater

Risk calculations were performed for residential adults and children exposed to groundwater although none of the groundwater is currently used for drinking water. Non-cancer hazards for adults exceeded 1.0 at 6.0. The majority of non-cancer hazard to the adult resident was the result of the inhalation of chloroform (HQ = 4.6). The exposure concentration (23 μg/L) was the maximum for this COPC; however, the exposure-point concentration is less than the MCL for total trihalomethanes of 100 μg/L. The non-cancer hazard for residential children was smaller than that for adults, but still exceeded 1.0 (cumulative HI = 3.6). These risks were from a number of metals, the largest of which was chromium with a HI of 1.4.

Cancer risks from exposure to chemicals (metals VOCs, and SVOCs) in the groundwater were above the acceptable cancer risk range (8×10^{-4}). Risks from benzo(a)pyrene and dibenz(a,h)anthracene contributed the majority of these risks at 3×10^{-4} and 2×10^{-4} respectively.

The results of the groundwater future residential adult and child risk assessment indicate that groundwater may not be appropriate for use as a potable water source.

6.4 Ecological Risk Assessment Summary

As reflected in HQs greater than the benchmark of 1.0, the Step 1 and 2 ERA screening process identified potential risk from a number of analytes for several receptors over several upland areas of the Fort Totten site. As identified in the Step 1 COPC-screening process, potential risk to lower trophic level organisms (i.e., plants and/or soil invertebrates) were identified for from 1 to 12 analytes, depending on the portion of the upland area of the Fort Totten FUDS examined. A number of the HQs were less than 10. However, some were greater than 10, indicating potential risk to lower trophic levels (i.e., aluminum, lead, mercury, and vanadium at both the fill area and other area [Tables 5-1 and 5-2]). Similarly, food-web calculations projected at least potential risk and in some cases the HQ was greater than 100.

The draft RI report, which contained an SLERA for the upland area, was reviewed by the State of New York, Department of Environmental Conservation, Division of Fish and Wildlife, the public, and the USACE project manager (i.e., the risk managers for the site). The SLERA in the draft RI report was updated with the soil data collected in 2004. In light of the potential risks, uncertainties, and conservative nature of the SLERA for the upland areas, the risk managers concluded, in the context of the SMDP process, that further efforts to define the risks to upland ecological receptors was not warranted. This initial conclusion is further supported by the additional soil data collected in 2004. USEPA guidance for ecological risk assessment (1997c) recognizes that because of the very conservative exposure assumptions used to identify COPCs, some of the identified chemicals may pose negligible risk. It is the conclusion of the SLERA that refinements of the identified hazards are not warranted.

6.5 Findings and Recommendations

The following findings and recommendations are based on the analysis and evaluation of 1) three rounds of soil data collected in the upland portion of the Ft. Totten FUDS, 2) indoor air samples from Building 615, and 3) multiple rounds of groundwater samples:

Findings

- Some metals and SVOCs are present in soil at the Other Area and Fill Area in the upland portion of the facility at concentrations that exceed state screening criteria.

- The sample results for the indoor air sampling in Building 615 indicated that there were no detectable concentrations of mercury greater than the state screening level.

- The sample results indicate that the groundwater at the FUDS may not be appropriate for use as a potable water source for residential consumption.

- It is the conclusion of the SLERA that refinement of the hazards identified are not warranted and that there are no unacceptable threats to ecological receptors from exposure to chemicals in the soil in the upland portions of the facility.

- Soil in the Other Area of the Ft. Totten FUDS does not present an unacceptable risk under the multiple scenarios evaluated and may be considered for unrestricted use.

- Soil in the Fill Area of the Ft. Totten FUDS presents an unacceptable hazard under a residential reuse scenario due to lead. Therefore, it may be in appropriate for future unrestricted reuse unless actions are taken to limit residential exposure. However, reuse of the site for nonresidential uses may be appropriate.

- The drain pipes located in Building 615 do not discharge into Little Bay and do not contain mercury at concentrations that would volatilize into the building and pose a health concern to workers.

Recommendations

- There is a suspected vault/drywell structure at the end of the floor drain in Building 615's photography shop. Recommend that the vault/drywell structure be investigated (and if necessary removed) and soil from that area be sampled and analyzed for mercury.

- Recommend a Focused Feasibility Study of the Fill Area soil.

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