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The Fort Totten mercury pollution risk assessment: A case history

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Abstract

Operational activities have resulted in mercury in the sediments surrounding Little Bay in Queens, NY. This is adjacent to Fort Totten, a formerly used defense site. Some of the mercury levels in these sediments exceeded New York State screening values. A human health risk assessment was accomplished, based on conservative assumptions. The risk assessment examined the potential for adverse health effects from direct contact with and ingestion of contaminated sediments/surface water and ingestion of biota. Potential exposures to recreational receptors including adults and children were examined. The highest numerical risk results from finfish ingestion and then second for exposure from dermal contact to the sediments. The only exposure pathway showing a hazard quotient greater than unity is finfish ingestion for the child. In summary, overall risk to this mercury exposure is minimal in spite of the state screening value being exceeded.
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Keywords: Mercury contamination; Sediments; Fish; Fort Totten; Little Bay; Risk assessment; Human health

1. Introduction

Pollution from toxic chemicals and their waste generates concern, because they affect human health, environment (water resources, air quality, and soil), ecology, and other non-living systems, such as buildings and other aesthetic resources. Other major issues of concern include bio-concentration in the food chain and persistence of a chemical in the environment. Management of hazardous chemicals and their disposal include both risk assessment and regulations to control these risks.

Mercury, in particular, generates much attention due to both its toxicity and complexity. Mercury exists in the vapor elemental form (Hg^0), an inorganic form (Hg^{+2}), and an organic compound in the form of methylmercury (CH_3Hg). Therefore, mercury can volatilize into the air sector, can bio-concentrate in fatty tissue and, thus, propagate in the food chain, and affects soils and sediments. Mercury is one of the persistent, bioaccumulative, and toxic chemicals under the U.S. Emergency Planning and Community Right-to-Know Act (EPCRA) [1].

Environmental risks are characterized as either human health or ecological or both [2,3]. This assessment focuses on human health risks. Some toxic chemicals show both carcinogenic and non-carcinogenic risks while others show one or the other

Abbreviations: AT, averaging time; BGS, below ground surface; BW, body weight; C, concentration; CDI, chronic daily intake; ED, exposure duration; EF, exposure frequency; ET, exposure time; HI, hazard index; HQ, hazard quotient; IR, intake rate; MRL, minimal risk level; ND, non-detect; PC, permeability constant; RfD, reference dose; RI, remedial investigation; RME, reasonable maximum exposure; SA, surface area; SVOC, semi-volatile organic compound; TCLP, toxicity characteristic leaching procedure; TDI, tolerable daily intake; UCL, upper confidence level (limit); USACE, United States Army Corps of Engineers; USCG, United States Coast Guard; USEPA, United States Environmental Protection Agency; USFDA, United States Food and Drug Administration; VOC, volatile organic compound

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[2]. Quantitative evaluation of health risks using quality data is essential to a thorough evaluation of the remedial options available to address contamination. The health effects of mercury depend on whether it is an inorganic or organic form, environmental conditions, mercury dose (strength of source), receptor vulnerability, and whether there is a complete exposure pathway [4]. Target organs for mercury include the respiratory system, skin, central nervous system, kidneys, and eyes [5]. Common symptoms from mercury exposure include: gastrointestinal, kidney, neurological, and respiratory disorders [4].

Fort Totten is a 147-acre (59.5 ha) site owned and operated by the U.S. Department of Defense since 1857 [6]. Until 1944, Fort Totten was used by the U.S. Army for national defense and engineer training. From 1944 and on, Fort Totten was operated by various U.S. Army commands, including a training center for U.S. Army reserves and engineers. Today, Fort Totten still functions as a training center, but the land composing Fort Totten is now owned by several other Federal agencies.

The installation is located in the Willets Point section of Queens County approximately 20 miles (32.2 km) east of New York, NY, USA. Fort Totten is located at the mouth of the East River in Queens, New York (north shore of Long Island) and is east of the southern reaches of the Throgs Neck Bridge near the confluence of Long Island Sound and the eastern entrance to the East River. The United States Coast Guard (USCG) property occupies the northwest portion of the Willets Point peninsula and is bounded by U.S. Army property on the north, east, and west. The 9.6-acre portion (3.89 ha) of Fort Totten operated by the USCG is the focus of this investigation. The installation layout is shown in Fig. 1.

The present work consists of a quantitative human health risk assessment of the mercury contamination in the marine sediments and biota of Little Bay, which is adjacent to Fort Totten. It describes the sampling locations, methods, analyses, and results. The objectives for this risk assessment is to determine the range of mercury contamination in the sediments, biota, and surface water in Little Bay and estimate risks to human health from residual contamination.

2. Previous investigations

The previous investigations are summarized in the U.S. Army Corps of Engineers (USACE) Remedial Investigation Report [6]. Mercury contamination was first discovered in Building 615 in April 1985, by the USCG. No mercury vapors were detected and a floor drain sediment sample was collected in May 1985. That sample was later analyzed with a group of seven sediment samples taken from Little Bay during an inspection in February 1986. The samples were analyzed for metals: arsenic, barium, cadmium, chromium, lead, mercury, selenium, and silver, by the Toxicity Characteristic Leachate Procedure (TCLP) [7]. The leachate for all the samples showed non-detect for all the metals, except the drain sediment, which contained 20–23% mercury. Subsequently, a 1988 Site Investigation was performed by USACE. Three sediment samples 6 in. (15.2 cm) below ground surface (BGS) were collected from Little Bay. Mercury was found at 1.5 mg/kg in a single sediment sample which exceeded New York State screening criteria of 0.15 mg/kg [8]. Thus, mercury is a chemical of potential concern.

In September and October 1989, the USCG collected and analyzed shoreline sediment samples adjacent to the Building 615 outfall, at 6 in. (15.2 cm), 10 in. (25.4 cm), and 24 in. (61.0 cm) BGS. The samples were analyzed for cadmium, chromium, copper, manganese, zinc, and lead using the TCLP. Mercury concentrations were arrayed in the shallow bay area near Building 615. The other metals were well below action levels, but mercury concentrations approximated action levels. However, TCLP testing indicated that mercury was not leaching: i.e., no leachable mercury was detected from any samples analyzed in this study.

In March 1995, USACE took samples from Building 615's floor drains and outfalls. These samples were analyzed for mercury and concentrations ranged from 0.111–0.206 mg/kg at the outfalls to 1.286 mg/kg at the western floor drain and 16.8 mg/kg at the eastern floor drain. The floor drains were removed in 1997 and the underlying soils tested for mercury. No mercury-contaminated soils were found.

In summary, the above-mentioned studies analyzed mercury contamination from Building 615 and its outfalls in Little Bay sediments. Mercury contamination around Building 615 and Little Bay varied from non-detect to 2.1 mg/kg.

3. Current investigation, sample collection, and results

Sampling activities are detailed in the RI Report for this site prepared by USACE [6]. Little Bay sediment, surface water, and biota were sampled as part of the Fort Totten investigation in early 1998. The most important state requirements for this project are from the New York State Department of Environmental Conservation (NYSDEC) [8–10].

3.1. Sediment sample collection

Initial sediment samples were collected to delineate mercury levels within the Little Bay in June 1998. Sediment samples at 16 additional locations were collected July 2000. The sediment sampling can be divided into four areas: (1) shoreline sediment sampling; (2) near shore sediment samples 0–50 ft (0–15 m) of the sea wall; (3) off shore sediment samples 50–400 ft (15–122 m) of the sea wall; and (4) other offsite sediment sampling. These areas are shown in Fig. 2. Each sample was subdivided into four sub-samples depending on sample depth BGS. The four sub-sample depth ranges were 0–6 in. (0–15 cm) BGS, 6–12 in. (15–30.5 cm) BGS, 12–18 in. (30.5–46 cm) BGS, and 18–24 in. (46–61 cm) BGS.

During the initial round of sampling (1998), samples deeper than 1 ft (30.5 cm) BGS generally were not available due to surficial rock outcroppings. This risk assessment uses the surface

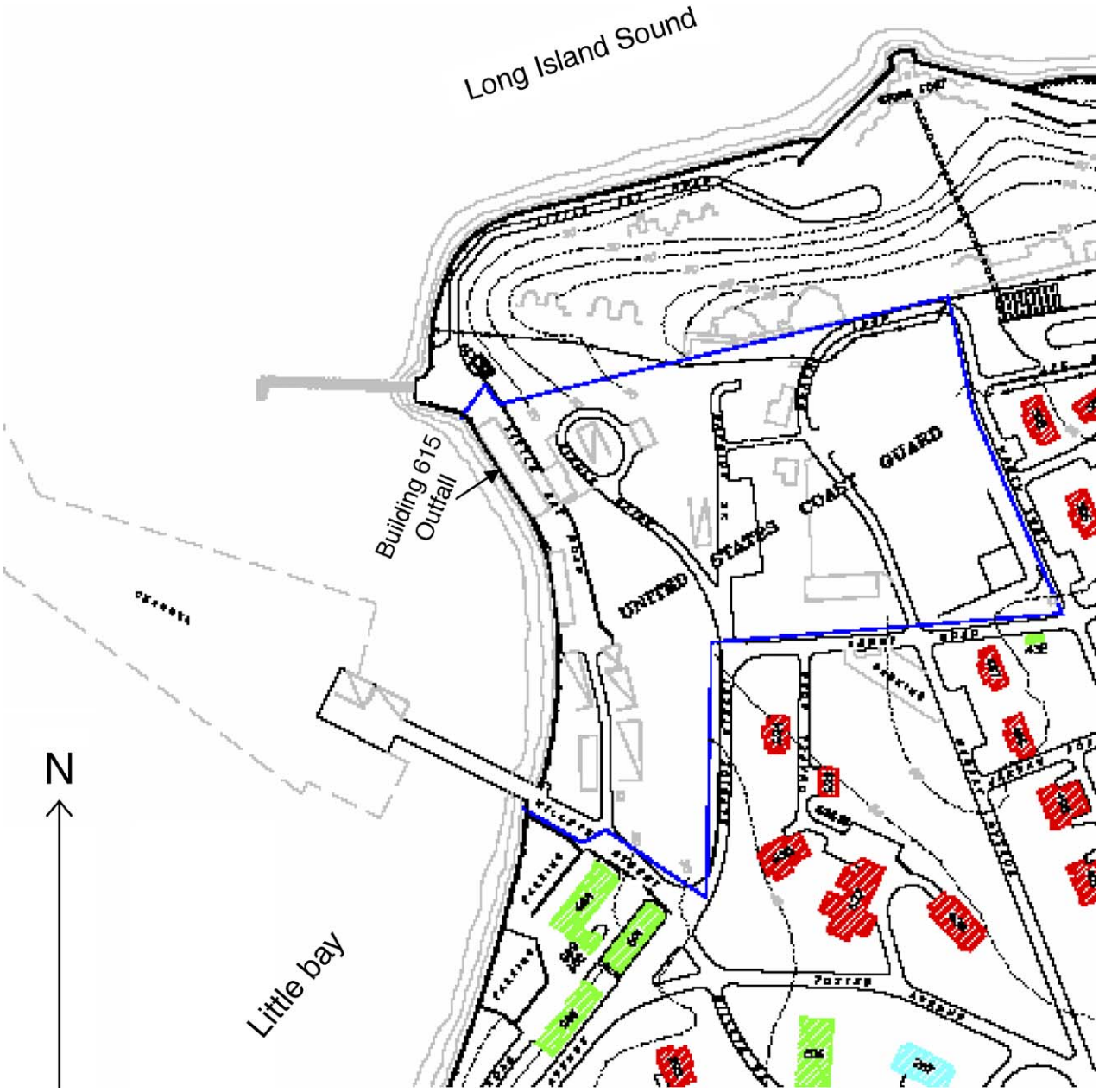

Fig. 1. Ft. Totten formerly used defense site installation map.

layer (0–1 ft BGS or 0–30.5 cm BGS) because the surface layer is the layer with which receptors would have contact. The later sampling (2000) expanded the sampling distance from seawall at Little Bay. These sub-sample splits were from 0 to 3 in. BGS (0–7.6 cm BGS), 3 to 12 in. BGS (7.6–30.5 cm BGS), and 1 to 2 ft BGS (30.5–61.0 cm BGS).

3.1.1. Shoreline sediment sampling

The seven shoreline sediment samples were evenly spaced and collected along the shoreline, with no more than 100 ft between samples. Mercury (Hg) levels range from 103 to 484 μg/kg 0–6 in. (0–15 cm) BGS of Little Bay and from 149 to 804 μg/kg 6–12 in. (15–30 cm) BGS. No samples were collected deeper than 12 in. (30 cm) BGS.

3.1.2. Near shore sediment sampling 0–50 ft (0–15 m) of sea wall

Thirty (30) sediment sample locations are within 50 ft (15 m) of the sea wall. A significant proportion of these close underwater sediment samples are located within Building 615's two outfall deltas. The outfall deltas are located by extending a line seaward (perpendicular to the sea wall), from the outfall's concrete cap at the sea wall. Outfall delta samples locations were taken at 1, 10, and 25 ft (0.3, 3.0, and 7.6 m) from the sea wall, and on the perpendicular axis, from each of these locations, two opposing transverse samples 10 ft (3.05 m) away from the perpendicular line. Therefore, there are 18 outfall delta samples, 9 from each outfall. Mercury levels range from 71.1 to 2850 μg/kg 0–6 in. (0–15 cm) BGS and from non-detect (ND)

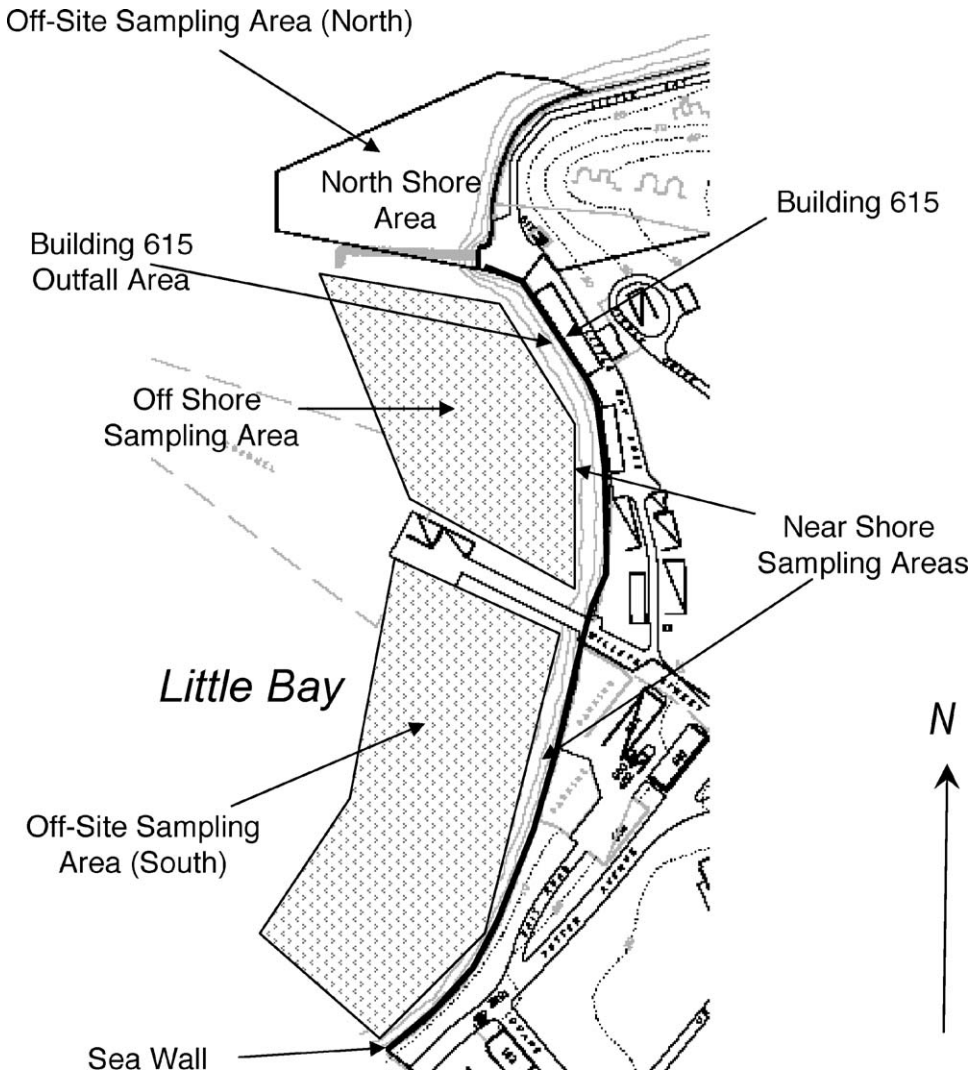

Fig. 2. Sediment sampling area.

to 5250 μg/kg 6–12 in. (15–30 cm) BGS. There were no samples collected deeper than 18 in. (46 cm) BGS and only one sample 12–18 in. (30–46 cm) BGS, where the mercury concentration was 504 μg/kg.

3.1.3. Off shore sediment sampling (50–400 ft (15–122 m) of sea wall)

There are 26 sediment samples locations, 50–400 ft (15–122 m) from the sea wall, approximately 100 ft apart, in a rectangular grid. The areas of sampling are shown in Fig. 2. Mercury levels range from ND to 1990 μg/kg 0–6 in. (0–15 cm) BGS and from ND to 2500 μg/kg 6–12 in. (15–30 cm) BGS.

3.1.4. Other offsite sediment sampling

There are 66 sediment sample locations (19 north and 47 south of Building 615), approximately 100 ft (30.5 m) apart in a rectangular grid starting from the shoreline. Mercury levels range from ND to 2030 μg/kg 0–6 in. (0–15 cm) BGS and from ND to 2520 μg/kg 6–12 in. (15–30 cm) BGS.

3.1.5. Confirmation sediment sampling

Sixteen (16) sediment samples were collected from portions of Little Bay, previously sampled, to confirm mercury concentrations measured in initial sampling events. The majority of these samples were collected at more than 300 ft (91.4 m) from the sea wall. Eight of the additional locations correspond to areas determined to contain some of the higher mercury sediment concentrations just west of Building 615 (between the northern rock jetty and Willets Street Pier). The other eight sediment sampling locations extended the area of sampling in regions of suspected elevated mercury levels, from 400 to 800 ft (121.9–243.8 m) from the sea wall. Previous sediment sampling had been within 400 ft (121.9 m) of the sea wall. Fig. 2 shows the areas that were sampled.

At each sampling location, 11 sediment sub-samples were collected: "surficial" (0–3 in. or 0–7.62 cm depth), 3–12 in. (7.62–30.5 cm), 1–2 ft (30.5–61 cm), 2–3 ft (61–91.44 cm), 3–4 ft (0.915–1.22 m), 4–5 ft (1.22–1.52 m), 5–6 ft (1.52–1.83 m), 6–7 ft (1.83–2.13 m), 7–8 ft (2.13–2.44 m), 8–9 ft (2.44–2.74 m), and 9–10 ft (2.74–3.05 m) BGS. A total of 176 sediment samples

were collected for chemical analysis, plus additional samples for quality assurance/quality control purposes, with 175 being analyzed for mercury (one broke during processing). Mercury levels range from 80 to 1200 μg/kg 0–3 in. (0–7.6 cm) BGS, ND to 2800 μg/kg 3–12 in. (7.6–30 cm) BGS, and from ND to 2800 μg/kg from 12–24 in. (30–61 cm) BGS.

3.2. Surface water sample collection

During two separate sampling events, a total of 24 surface water samples (12 at the top of the water column and 12 at the bottom of the water column) were collected within 50 ft (15.2 m) of the sea wall. The surface water and bottom water samples were taken as a pair; wherever a surface water sample was taken, a bottom water sample was taken below it. All samples were taken within 50 ft (15.2 m) of the sea wall at high tide. The water samples were collected within 0.5 m of the surface and bottom. Virtually, all sample results were at or below the reporting limit of 0.1 μg/L. The highest concentration of 0.27 μg/L showed a result above the instrument detection limit but below the contract-required detection limit. This is at the low end of quantitation limits so this quantity is estimated. For this risk assessment, the maximum water concentration of 0.27 μg/L is used. This represents a very conservative assessment of the risk of mercury exposure for surface water ingestion and dermal contact.

3.3. Biota sample collection

The mussel (*Mytilus edulis*) and oyster (*Crassostrea virginica*) samples were taken within 50 ft (15.2 m) of the sea wall. Approximately half the samples were collected from the deltas of the two Building 615 outfalls and the other half from relatively evenly spaced locations, within 50 ft (15.2 m) of sea wall. Five (5) mussels and 9 oysters were collected. Fish were collected between the pier and the breakwater. Seven (7) windowpane flounder (*Scophthalmus aquosus*), 10 mummichogs (*Fundulus heteroclitus*), 5 juvenile striped bass (*Morone saxatilis*), and 5 white flounder (*Plueronectes americanus*) were caught. Both filet and whole body samples were collected and analyzed. All biota samples were analyzed only for mercury and results are expressed in wet weight.

All 5 mussel and 9 oyster sample results and 4 of 5 blue crab (*Callinectes sapidus*) sample results were ND. The reporting limit ranges from 0.08 to 0.11 mg/kg and the maximum concentration observed is 0.10 mg/kg for the shellfish. All juvenile striped bass (*Morone saxatilis*) sample results, 9 of 10 mummichogs (*Fundulus heteroclitus*) sample results, 3 of 5 (including both filet and whole body) white flounder sample results, and 1 of 7 (including both filet and whole body) windowpane flounder sample results were ND. The reporting limit ranges from 0.05 to 0.10 mg/kg and the maximum concentration observed is 0.27 mg/kg from the windowpane flounder. The maximum finfish concentration was 0.27 mg/kg.

3.4. Sample analysis

All sediment and biota samples were analyzed by USEPA Solid Waste Method 7471 (cold-vapor atomic absorption method based on the absorption of radiation at the 253.7 nm wavelength by mercury vapor and involves an acid digestion preparatory step). USEPA Solid Waste Method 7470, also a cold-vapor atomic absorption method, was used to analyze all surface water samples. The analytical method reduces the mercury to an elemental state and aerates it from solution in a closed system. The mercury vapor passes through a cell positioned in the light path of an atomic absorption spectrophotometer. Absorbance (peak height) is measured as a linear function of mercury concentration. Quality control and quality assurance samples were collected at a rate of 10% of all samples by media. All sample results were within method specific control limits.

4. Conceptual site model

The first step in evaluating exposure is the development of a conceptual site model. The basic elements of a conceptual site model are described in USEPA [2]. The goal of developing the conceptual site model is to characterize the site with respect to its physical characteristics as well as those of the human populations on and near the site. The output of this step is a qualitative evaluation of the site and surrounding populations with respect to those characteristics that influence exposure. All information gathered during this step will support the identification of complete plausible exposure pathways. In addition, the information on the potentially exposed populations is used to determine the values of some intake variables.

4.1. Environmental setting and contaminants at the site

Building 615 was originally used as a torpedo and mine repair facility. The armaments contained mercury in their guidance systems and when repair required mercury removal, it was disposed of through the floor drains. Therefore, Building 615 is a source of the mercury contamination and is adjacent to Little Bay, a small inlet on the greater New York City harbor area. The site is located in the Willets Point section of Queens County, near Bayside, NY (north shore of Long Island). It is located at the confluence of the Long Island Sound and the eastern entrance to the East River. Current use of this site is restricted. Future reuse of the site and shoreline perimeter includes a public esplanade along the entire length of Little Bay. The site will be reserved as open space/recreational property. The esplanade is a proposed continuous multi-use path at the peninsula's edge.

4.2. Contaminant fate and transport

A conceptual site model for this site is shown in Fig. 3. The source of the mercury contamination is the floor drain in Building 615. From this drain mercury entered Little Bay and came in contact with the surface water and sediment. The mercury released was most likely elemental mercury used as switches in torpedoes and mines. Based on its chemical properties most of the mercury would have initially partitioned into the sediment due to the elemental mercury's low water solubility.

Mercury is a metal with atomic number 80, atomic weight 200.59, and density 13.5 g/cm^3. At ambient conditions, mercury

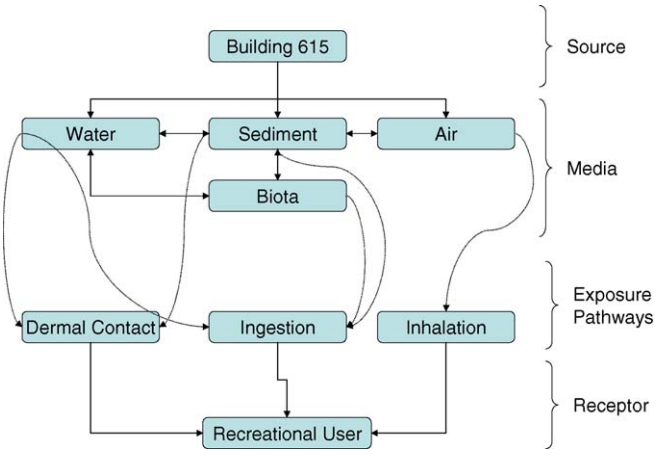

Fig. 3. Conceptual site model for risk assessment.

is a liquid metal in its elemental (inorganic) form. Mercury may be present in the environment in three forms: elemental, organic, and inorganic. Mercury can exist in three oxidation states: Hg^0 (metallic), Hg_2^{2+} (mercurous), and Hg^{2+} (mercuric-Hg(II)). The properties and chemical behavior of mercury strongly depend on the oxidation state. Mercurous and mercuric mercury can form numerous inorganic and organic chemical compounds; however, mercurous mercury is rarely stable under ordinary environmental conditions. Mercury is unusual among metals because it may form covalent rather than ionic bonds. Most of the mercury encountered in water/soil/sediments/biota (all environmental media except the atmosphere) is in the form of inorganic mercuric salts and organomercurics. Organomercurics are defined by the presence of a covalent carbon–Hg bond. The presence of a covalent carbon–Hg bond differentiates organomercurics from inorganic mercury compounds that merely associate with the organic material in the environment but do not have the carbon–Hg bond. The compounds most likely to be found under environmental conditions are these: the mercuric salts $HgCl_2$, $Hg(OH)_2$, and HgS; the methylmercury compounds, methylmercuric chloride (CH_3 HgCl), and methylmercuric hydroxide (CH_3 $HgOH_2$); and, in small fractions, other organomercurics (i.e., dimethylmercury and phenylmercury) [11]. Mercury compounds in the aqueous phase often remain as undisassociated molecules, and the reported solubility values reflect this. Solubility values for mercury compounds that do not disassociate are not based on the ionic product. Most organomercurics are not soluble and do not react with weak acids or bases due to the low affinity of the mercury for oxygen bonded to carbon. CH_3 HgOH, however, is highly soluble due to the strong hydrogen bonding capability of the hydroxide group. The mercuric salts vary widely in solubility. For example, $HgCl_2$ is readily soluble in water and HgS is as unreactive as the organomercurics due to the high affinity of mercury for sulfur. The dominant form in the atmosphere is vapor-phase elemental mercury.

Geochemical cycling caused by biotic and abiotic process would then cycle the mercury though several different chemical forms (e.g., elemental, organic, and inorganic). The USEPA Report to Congress [11] contains an overview of the mercury geochemical cycle from a global and regional perspective and a detailed description. Briefly, most mercury in water, soil, sediments, or plants and animals is in the form of inorganic mercury salts and organic forms of mercury (e.g., methylmercury). The inorganic form of mercury, when bound to airborne particles or in a gaseous form, is readily removed from the atmosphere by precipitation and is also dry deposited. Wet deposition is the primary mechanism for transporting mercury from the atmosphere to surface waters and land. Even after it deposits, mercury commonly is emitted back into the atmosphere as a gas or associated with particles, to be re-deposited elsewhere. As mercury cycles between the atmosphere, land, and water, mercury undergoes a series of complex chemical and physical transformations, many not fully understood [11]. After partitioning into the several different media, some degree of equilibrium would be achieved. The mercury continues to move through the different media and is influenced by physical disturbances and its chemical oxidation state. Due to the type of release, the media most impacted are the sediment and surface water.

Once released into the bay from drainpipes at Building 615, the mercury was deposited in the sediment. The geochemical cycling of mercury in the bay would include chemical transformation from elemental mercury, to inorganic and organic mercury. Mercury can exist in three oxidation states: Hg^0 (metallic), Hg_2^{2+} (mercurous), and Hg^{2+} (mercuric). The physical and chemical properties and toxic effects of mercury depend on the oxidation state.

Mercury can enter surface water as Hg^0, Hg^{2+}, or methylmercury. Once in aquatic systems, mercury can exist in dissolved or particulate forms and can undergo the following transformations.

- Hg^0 in surface waters can be oxidized to Hg^{2+} or volatilized to the atmosphere.
- Hg^{2+} can be methylated in sediments and the water column to form methylmercury.
- Methylmercury can be alkylated to form dimethylmercury.
- Hg^{2+} and methylmercury can form organic and inorganic complexes with sediment and suspended particulate matter.

Each of these reactions can also occur in the reverse direction. The net rate of production of each mercury species is determined by the balance between forward and reverse reactions. Estimates of the percent of total mercury in surface waters that exists as methylmercury vary. Generally, methylmercury makes up less than 20 % of total mercury in the water column [4].

5. Human health risk assessment

The range of mercury contamination in various media was determined through extensive sampling. This section quantifies human health risks from exposure to mercury under a recreational exposure scenario.

5.1. Exposure assessment

The exposure assessment uses the conceptual site model, described previously, to quantify the relationship between the

media containing the mercury and the receptor. An integral part of the chemical–receptor interaction is the planned reuse of the site. The first step in the exposure assessment is to qualitatively describe each receptor for the pathways identified in the conceptual site model. The types of receptors will be based on future land use at this site. The second step is to quantify the exposure point concentration of mercury in each media of interest. The third step is to quantify the characteristics of the receptor that impact exposure. This last step is performed for each pathway individually.

Relative to Building 615 and Little Bay, the planned waterfront esplanade is a proposed continuous multi-use path at the peninsula's edge [12]. The esplanade will be developed at the water's edge where possible and will move inland to preserve existing vegetation or other site features.

5.1.1. Exposure pathways

Access to the site currently is restricted. There is no receptor for the portion of the Little Bay where the mercury is located. While there may be occasional personnel in Building 615, no receptors are on the shoreline with a reasonably quantifiable frequency.

The future receptors evaluated include an adult and child recreational angler/beach comber. This scenario is consistent with the approved redevelopment plan [12]. The future receptors are assumed to recreate along the shoreline including fishing, collecting shellfish, collecting shoreline items, and occasional wading. Swimming was not considered likely because of the rocky and uninviting nature of the shoreline in the area of Building 615.

There are three possible exposure pathways for the future receptors: inhalation, ingestion, and dermal contact. Ingestion and dermal contact were evaluated for each of the two receptors. While inhalation is a complete pathway, it was not assessed quantitatively because the sediments are underwater for approximately 12 h a day and particle size and soil/sediment moisture content is such that fugitive dust emissions are unlikely. Hence, only ingestion and dermal contact were quantitatively evaluated. The pathways evaluated include ingestion of water, biota, and sediment, and dermal contact with water and sediment.

5.1.2. Exposure point concentration

The mercury concentrations in the sediment, water, and biota were determined with a sampling and analysis effort. Statistics were used to characterize the distribution of mercury concentrations. The numbers and types of samples collected are discussed earlier. The exposure point concentrations were calculated following USEPA Guidance [2].

All of the surface water samples collected were used to determine the exposure point concentration. This included samples from the top of the water column and those collected near the sediment/water interface. Due to the large number of non-detected values and the small dataset, the maximum observed concentration was used as the exposure point concentration (0.00027 mg/L).

To evaluate human receptor exposure to mercury in sediment, the samples collected within 50 ft (15.2 m) of the shoreline were used. All samples collected from 0 to 6 in. (0–15.2 cm) and 6 to 12 in. (15.2–30.5 cm) were combined into one dataset because the exposure unit depth of 12 in. (30.5 cm) was assumed. Only samples in the surface layer (0–1 ft BGS or 0–30.48 cm BGS) were used for this sediment risk assessment, as it is the surface layer likely at which exposure would occur. To determine the exposure point concentration, the USEPA computer program ProUCL Version 3.0 was used to determine the distribution and calculate upper confidence limit of the mean [13,14]. Non-detected sample results were assigned a random number between 0 and the analytical detection limit (0.050 mg/kg). The exposure point concentration for this non-parametric date set was the 95% Chebyshev estimate of the mean (0.840 mg/kg).

All biota samples collected were used to calculate the mercury concentration in finfish and shellfish. The data from all finfish samples were combined; likewise, data for all shellfish samples were combined. There were too few samples and too many non-detects in each species data subset to determine the distribution for each species. The maximum reported concentration of mercury on each type was used as the exposure point concentration. The exposure point concentrations were 0.27 mg/kg for finfish and 0.10 mg/kg for the shellfish.

5.1.3. Characteristics of exposure

There are no current human receptors with a quantifiable frequency because Ft. Totten is a closed military installation and site access is controlled. Future receptors could include adults and children who recreate on the beach. The relevant exposure characteristics for the future adult and child receptor are summarized in Table 1.

Exposure characteristics labeled as "site-specific" were estimated conservatively based on anticipated exposures and comparisons to data from other sources [15,16]. The characteristics are based on the exposure profile for the reasonable maximum exposure (RME). The RME scenario is an attempt to describe exposures at the upper percentiles (e.g., 90th–95th) of the exposure profile [2,14]. The intake values used are from USEPA [15] and represents marine recreational anglers in the U.S. mid-Atlantic region.

The chronic daily intake (CDI) for incidental ingestion of sediment is an event-based value rather than a daily rate. The daily rate, provided by USEPA [15], for residential receptors is 100 mg/day for adults and 200 mg/day for children. An event-based rate was calculated to better reflect the event (i.e., episodic) nature of beach combing. As discussed in Chapter 1 of the USEPA Exposure Factors Handbook [15], it is important to define the duration estimate so that it is consistent with the intake rate. The objective is to define the terms so that when multiplied, they provide the appropriate estimate of mass of chemical contacted. Weighting the USEPA supplied values by an event duration of 2 h out of a possible 16 h per day, the intake rate per event for adults is 12.5 mg/event and for children it is 50 mg/event.

5.1.4. Estimated exposure profile

Using the characteristics of the exposed receptors and the exposure point concentrations calculated earlier, the CDI for

Table 1
Exposure parameters for adult and child future receptors, Little Bay, Ft. Totten, Queens, NY, USA

Exposure characteristic	Variable	Value	Units	Source
Incidental surface water ingestion				
Concentration in surface water	C_SW	0.00027	mg/L	Site-specific
Intake rate, adult	SW_IR_a	0.05	L/h	USEPA (1989)
Intake rate, child	SW_IR_c	0.1	L/h	USEPA (1989)
Exposure frequency, adult	SW_EF_a	52	Events/year	Site-specific
Exposure frequency, child	SW_EF_c	52	Events/year	Site-specific
Exposure time, adult	SW_ET_a	2	h/event	Site-specific
Exposure time, child	SW_ET_c	4	h/event	Site-specific
Dermal absorption from surface water				
Concentration in surface water	C_SW	0.00027	mg/L	Site-specific
Surface area available for contact, adult	SWD_SA_a	20000	cm^2/event	USEPA (1997)
Surface area available for contact, child	SWD_SA_c	7300	cm^2/event	USEPA (1992)
Skin permeability constant, adult	SWD_PC_a	1.00E−03	cm/h	USEPA (1992)
Skin permeability constant, child	SWD_PC_c	1.00E−03	cm/h	USEPA (1992)
Exposure time, adult	SWD_ET_a	2	h/day	Site-specific
Exposure time, child	SWD_ET_c	4	h/day	Site-specific
Exposure frequency, adult	SWD_EF_a	52	Days/year	Site-specific
Exposure frequency, child	SWD_EF_c	52	Days/year	Site-specific
Ingestion of finfish/shellfish				
Concentration in finfish/shellfish	C_Fish	0.27/0.10	mg/kg	Site-specific
Intake rate, adult	Fish_IR_a	0.0189/0.013	kg/day	USEPA (1997)
Intake rate, child	Fish_IR_c	0.009/0.007	kg/day	USEPA (1997)
Exposure frequency, adult	Fish_EF_a	365	Days	USEPA (1997)
Exposure frequency, child	Fish_EF_c	365	Days	USEPA (1997)
Incidental sediment ingestion				
Concentration in sediment	C_Sed	0.683	mg/kg	Site-specific
Intake rate, adult	Sed_IR_a	12.5	mg/event	Site-specific
Intake rate, child	Sed_IR_c	50	mg/event	Site-specific
Exposure frequency, adult	Sed_EF_a	52	Events/year	Site-specific
Exposure frequency, child	Sed_EF_c	52	Events/year	Site-specific
Sediment dermal exposure				
Concentration in sediment	C_Sed	0.683	mg/kg	Site-specific
Surface area available for contact, adult	Sed_D_SA_a	5800	cm sq/event	USEPA (1992)
Surface area available for contact, child	Sed_D_SA_c	2327	cm sq/event	USEPA (1992)
Sediment/skin adherence factor, adult	Sed_D_AF_a	1.00	mg/cm^2	USEPA (1992)
Sediment/skin adherence factor, child	Sed_D_AF_c	1.00	mg/cm^2	USEPA (1992)
Skin absorption constant, adult	Sed_D_ABS_a	1.0	Unitless	
Skin absorption constant, child	Sed_D_ABS_c	1.0	Unitless	
Exposure frequency, adult	Sed_D_EF_a	52	Events/year	Site-specific
Exposure frequency, child	Sed_D_EF_c	52	Events/year	Site-specific
Common variables				
Exposure duration, adult	ED_a	30	Years	USEPA (1989)
Exposure duration, child	ED_c	6	Years	USEPA (1989)
Body weight, adult	BW_a	70.0	kg	USEPA (1997)
Body weight, child	BW_c	19.7	kg	USEPA (1997)
Averaging time, adult	AT_nc_a	10950	Days	USEPA (1997)
Averaging time, child	AT_nc_c	2190	Days	USEPA (1989)
Event-driven ingestion rate				
Daily rate, adult	DR_a	100	mg/day	USEPA (1997)
Event duration, adult	EvD_a	2	h	Site-specific
Conversion factor	CF	16	h/day	Site-specific
Ingestion per event, adult	I-E_a	12.5	mg/event	Calculated
Daily rate, child	DR_c	200	mg/day	USEPA (1997)
Event duration, child	EvD_c	4	h	Site-specific
Conversion factor	CF	16	h/day	Site-specific
Ingestion per event, child	I-E_c	50	mg/event	Calculated

each receptor was calculated. The generic CDI equation for each exposure pathway is as follows:

$$\text{CDI} = \frac{C \times \text{IR} \times \text{ED}}{\text{BW} \times \text{AT}}$$

where CDI is equal to total potential dose, i.e., the product of the exposure point concentration (C), intake rate (IR), and exposure duration (ED) divided by the product of the body weight (BW) and averaging time (AT) [2]. From this generic equation, several pathway-specific equations were derived. The dose is expressed in mg of chemical per kg of body weight per day, mg/(kg day). These intakes are representative of the RME scenario and the average or median exposure would be less. The compounding of several upper percentile exposure estimates results in total pathway exposure that may approach or exceed the 99.99th percentile exposure [17].

For both surface water and sediments, the exposure pathways include ingestion and dermal contact, whereas for biota (shellfish and finfish), ingestion is the only exposure pathway. The CDI for each exposure pathway was calculated as follows:

$$\text{Surface water ingestion CDI} = \frac{(C)(\text{IR})(\text{ET})(\text{EF})(\text{ED})}{(\text{BW})(\text{AT})}$$

$$\text{Surface water dermal contact CDI} = \frac{(C)(\text{SA})(\text{PC})(\text{VCF})(\text{ET})(\text{EF})(\text{ED})}{(\text{BW})(\text{AT})}$$

$$\text{Biota ingestion CDI} = \frac{(C)(\text{IR})(\text{EF})(\text{ED})}{(\text{BW})(\text{AT})}$$

$$\text{Sediment ingestion CDI} = \frac{(C)(\text{IR})(\text{ET})(\text{EF})(\text{ED})(\text{MCF})}{(\text{BW})(\text{AT})}$$

$$\text{Sediment dermal contact CDI} = \frac{(C)(\text{SA})(\text{AF})(\text{ABS})(\text{EF})(\text{ED})(\text{MCF})}{(\text{BW})(\text{AT})}$$

where EF is the exposure frequency (events), ET the exposure time (h), PC the permeability constant for skin (cm/h), AF the adherence factor (mg/cm^2), SA the surface area (cm^2), and ABS is the skin absorption constant = 1, which assumes complete absorption through the skin for both adult and child. The two conversion factors used in the above CDI equations are as follows:

$$\text{Volume conversion factor (VCF)} = 0.001\ \text{L/cm}^3$$

$$\text{Mass conversion factor (MCF)} = 10^{-6}\ \text{kg/mg}$$

As noted earlier, the shellfish include the mussels, oysters, and the blue crab, whereas the finfish include the windowpane flounder, white flounder, mummichogs, and juvenile striped bass.

Chronic daily intakes for adult exposure ranges from a low of 2.14E−08 mg/(kg day) to a high of 7.29E−05 mg/(kg day) and for child exposure from a low of 5.70E−08 mg/(kg day) to a high of 1.23E−04 mg/(kg day). In both the adult and child exposure scenarios, the pathway resulting in the highest CDI intake is the ingestion of finfish and shellfish.

5.2. Toxicity assessment

There are toxicity data for three forms of mercury (elemental, inorganic, and methylated) that may be present at this site. Detailed reviews of the toxicity of mercury are contained in the recent Agency for Toxic Substances and Disease Registry's Toxicity Profile for Mercury [4], USEPA Integrated Risk Information System [5], and USEPA Report to Congress, 1997 [11].

5.2.1. Quantitative cancer endpoints

It is not certain whether exposure to the various forms of mercury can cause cancer [4,5]. USEPA has determined that elemental mercury should be categorized as "D" not classifiable with regards to carcinogenic potential. Inorganic and organic mercury are categorized as "C"—possible human carcinogens. USEPA classifies methylmercury as group "C" based on inadequate data in humans and increased incidence of kidney tumors in a single species and sex [5]. Mice exposed to methylmercuric chloride in the diet had an increased incidence of kidney tumors [4,5]. The kidney epithelial cell tumors were observed only in the presence of profound nephrotoxicity and may be a consequence of cellular repair. Several non-positive cancer bioassays were also reported. Although genotoxicity data suggest that methylmercury may produce chromosomal and nuclear damage, there are also non-positive genotoxicity data. Cancer slope factors for all forms of mercury are unavailable, therefore a quantitative statement of cancer risk cannot be made at this time.

5.2.2. Quantitative non-cancer endpoints

Data in both humans and experimental animals show that all three forms of mercury (elemental, inorganic, and methylmercury) can produce adverse health effects at sufficiently high doses. Human exposure to elemental mercury occurs in some occupations, and exposure to inorganic mercury can arise from mercury amalgams used in dental restorative materials. Like all chemicals, mercury can produce a variety of adverse effects, depending on the dose and time of exposure. In general, mercury adversely affects the central nervous system. Health endpoints other than neurotoxicity were evaluated by USEPA [4,5].

The USEPA has developed Reference Doses (RfD) to determine safe levels of chemical exposure [2]. An RfD is defined by USEPA as "an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime" [18]. There are RfDs for methylmercury and mercuric chloride; however, there is no RfD for elemental mercury [5]. The available RfD for methylmercury is 0.0001 mg/(kg day) with an uncertainty factor of 10 is based on the critical endpoint of development of neurologic abnormalities in human infants. The available RfD for mercuric chloride is 0.001 mg/(kg day) with an uncertainty factor of 1000 is based on the critical endpoint of autoimmune effects in subchronic rat feeding studies.

Other U.S. federal agencies have also examined the health effects of mercury exposure. The U.S. Food and Drug Administration (USFDA) uses an action level based on consideration of the tolerable daily intake (TDI) for methylmercury, as well as information on seafood consumption and associated exposure to methylmercury [19]. The TDI is the amount of methylmercury that can be consumed daily over a long time with a reasonable certainty of no harm to adults. The neurological endpoint evaluated was paresthesia. USFDA in cooperation with the World Health Organization (WHO) established a TDI based on a weekly tolerance of 0.3 mg of total mercury per person, of which no more than 0.2 mg should be present as methylmercury. These amounts are equivalent to 5 and 3.3 μg, respectively, per kilogram of body weight. Using the values for methylmercury, this tolerable level would correspond to approximately 230 μg/week for a 70 kg person or 0.0004 mg/(kg day). Therefore, the USFDA's tolerable intake level is higher than that used by USEPA.

The U.S. Agency for Toxic Substance and Disease Registry (ATSDR) also developed a benchmark for mercury [4]. When calculated for exposure via ingestion, the minimal risk level (MRL) is conceptually equivalent to the RfD and the TDI. The MRL is calculated to ensure a substantial margin of safety. The MRL is not a definitive line indicating the boundary between no health risk and a definitive health risk. In 1994, ATSDR published a draft MRL for ingested methylmercury of 0.0001 mg/(kg day) (equivalent to the RfD for methylmercury). After re-evaluation of the data, a long public comment period and incorporation of additional toxicological studies, ATSDR revised the draft MRL for a final value of 0.0003 mg/(kg day) [20].

The available health-based benchmarks for mercury range from 0.0001 to 0.001 mg/(kg day). However, the health-based benchmarks for methylmercury fall within a narrower range of 0.0001–0.0004 mg/(kg day). When making quantitative estimates of non-cancer hazards from mercury exposure, the methylmercury RfD developed by USEPA is used. Specifically, the RfD for methylmercury is used because the sampling program was not designed to differentiate between elemental, organic, and inorganic mercury. This approach is consistent with observations that most (>95%) of the total mercury content of fresh and saltwater fish is methylmercury [11]. In addition, because mercury was not speciated in sediment or surface water samples, it was likewise assumed that all mercury present was methylmercury. This assumption will tend to overestimate the toxicity of mercury. By using the RfD for methylmercury, the toxicity assessment takes a conservative approach to estimating the potential health hazard from exposure.

5.3. Risk characterization

Risk characterization combines toxicity and exposure information to make a quantitative statement on the hazards and risks posed by the chemical of concern [2]. Risk characterization summarizes key issues and conclusions of each of the other components of the risk assessment and describes the likelihood of harm. Included in the summary is a description of the overall strengths and limitations (including uncertainties) of the assessment and conclusions. The risk characterization includes, at least in a qualitative sense, a discussion of how a specific risk and its context compares with other similar risks.

5.3.1. Risk/hazard profile

Combining toxicity data (RfD = 0.0001 mg/(kg day)) with exposure data (CDI in mg/(kg day)), the hazard quotient (HQ) is calculated for each exposure pathway using the general formula:

$$\mathrm{HQ} = \frac{\mathrm{CDI}}{\mathrm{RfD}}$$

The resulting HQ is a unitless number that represents the ratio of the estimated dose from exposure at the site to the RfD, which is assumed to be without adverse health impacts. The HQ is not a probability of harm and HQ = 0.01 does not mean that there is a one in one hundred chance of the adverse effect occurring. Likewise, HQ > 1 one does mean that adverse effects will or have occurred, but that adverse effects would be expected based on the exposure scenario and toxicity data presented. Inherent in any HQ are several uncertainties that should be evaluated prior to making a definitive conclusion.

Since the HQ's < 1 for each pathway, except finfish ingestion, adverse health effects are not expected to result from the exposures described in the assessment. Actual exposure from each pathway will probably be less than that estimated. The CDI and HQ for each exposure pathway and receptor are summarized in Table 2.

A total exposure hazard index (HI) is calculated, by summing the individual pathways. The HI represents the hazard posed by exposure to mercury from all routes of exposure. The HI for the adult receptor is 1.02 and for the child the HI is 1.74. Because the HI for the adult is approximately unity, no adverse health effects are expected to result from the total exposures estimated in view of the conservative assumptions. Like the HQ, the HI is a unitless number that is not a probability of harm. The HI should not be interpreted as a bright line standard below which no effects will occur and above which effects will occur. The HI should be examined in light of the uncertainties and assumptions in the entire risk assessment.

5.3.2. Uncertainties in the risk assessment

Uncertainties are inherent in any risk assessment. Uncertainties can be broken into three separate areas: sample collection/analysis, exposure assessment, and toxicity assessment [2]. Within each area, uncertainties can be site-specific or generic. Site-specific uncertainties are influenced by site conditions. Generic uncertainties are outside the influence of the site, e.g., laboratory analysis of sediment samples is dependent on the calibration of the analytical instrument. Care is exercised in all areas to limit the uncertainties, but all uncertainties will never be completely eliminated.

During sample collection, some samples were purposefully collected from areas of known chemical contamination. This

Table 2
Chronic intake and hazard quotients for multiple exposure pathways, Little Bay, Ft. Totten, Queens, NY, USA

Exposure pathway	CDI (mg/(kg day))	HQ
Surface water ingestion		
Adult	5.4951×10^{-8}	$0.00055 \ll 1$
Child	7.8107×10^{-7}	$0.0078 \ll 1$
Surface water dermal contact		
Adult	2.198×10^{-8}	$0.00022 \ll 1$
Child	5.7015×10^{-8}	$0.00057 \ll 1$
Shellfish ingestion		
Adult	1.8571×10^{-5}	$0.19 < 1$
Child	3.5533×10^{-5}	$0.36 < 1$
Finfish ingestion		
Adult	7.2900×10^{-5}	$0.73 < 1$
Child	1.2335×10^{-4}	$1.23 > 1$
Sediment ingestion		
Adult	2.137×10^{-8}	$0.00021 \ll 1$
Child	3.0373×10^{-7}	$0.0030 \ll 1$
Sediment dermal contact		
Adult	9.915×10^{-6}	$0.099 < 1$
Child	1.4136×10^{-5}	$0.14 < 1$
Total over all exposure pathways		
Adult	1.015×10^{-4}	$1.02 > 1$
Child	1.743×10^{-4}	$1.74 > 1$

action biases the results to reflect areas of higher contamination rather than have equal representation across the entire exposure area. More sediment, and mussel/oyster samples were taken near Building 615's outfall, than anywhere else. This results in a higher exposure point concentration than would be encountered if sampling were performed at random.

The exposure assessment tends to be conservative and overestimate the actual exposure of any specific individual. Most of this conservatism results from multiplying a series of upper-percentile exposure estimates together to estimate a reasonable maximum [17]. This approach is the result of USEPA policy to be conservative and protective of human health [2].

The toxicity assessment is not site-specific because it does not account for the type of mercury present in the exposure media. This uncertainty is a direct result of how the sampling and analysis plan was executed. Because the chemical form of mercury was undetermined, the health benchmark for the most toxic form of mercury was used, i.e., methylmercury. It was assumed that all mercury was present in the water column as methylmercury, while usually only 20% of the mercury in the water column is present as methylmercury [11]. While it is unlikely that all of the mercury present in all samples is methylated, faced with a lack of data the toxicity assessment uses a conservative, health protective approach.

Finally, using the maximum observed sediment mercury level (5.25 mg/kg) shows a hazard index of indicating mild risk for the adult (HI = 1.54), and a somewhat elevated risk for the child (HI = 2.50). Both of the sediment HQ's (ingestion and dermal contact) are still each individually under 1. However, the mercury level of 5.25 mg/kg is 6–12 in. (15–30 cm) BGS, and is only one sample. Using the maximum concentration of the mercury in the sediment of 2.85 mg/kg in the surface layer, which is still just one point of over 200 sample points in this layer, the adult HI = 1.25, i.e., the risk to the adult is borderline. However, the risk to the child using 2.85 mg/kg is still elevated (HI = 2.09) but less than the maximum sediment mercury level. Using the more likely, but still conservative assumption of the 95% UCL, of mercury concentration in 0–6 in. (0–15 cm) BGS samples, gives a child's hazard index of 1.74 and the adult HI approximately one. Furthermore, most of the higher mercury concentration in the samples collected from 0 to 6 in. (0–15 cm) BGS are more than 100 ft (30.5 m) out from the shoreline, an area not like to be contacted, because it is underwater.

5.4. Human health risk assessment summary

Activities at Building 615 resulted in mercury release to the environment. As mentioned previously, mercury can exist in the environment as organic, inorganic and elemental and toxicity depends on the form encountered. Exposure to mercury could occur along the shoreline below Building 615 when the property is transferred to public control. Current exposures are so sporadic that they are non-quantifiable. Future potential exposure scenarios include fishing, wading, and beach combing. Exposure to contaminated media might occur by incidental ingestion of sediment and/or surface water, dermal contact with sediments and/or surface water, and ingestion of finfish/shellfish. Ratios of high-end exposure and sensitive toxicity benchmarks indicated that adverse health effects are not expected. Adverse health effects from exposure to the contaminated media near Building 615 is not anticipated, based on the total HI. Overall, human health risk is expected to be minimal for the adult and slightly elevated for the child from this mercury exposure. The slightly elevated risk for the child results mainly from the consumption of finfish. However, fish from Little Bay may pose other risks due to other industrial pollution from this general area, e.g., from polychlorinated biphenyls (PCBs) or other hazardous organic chemicals that tend to magnify up the food chain. New York State currently has a fishing advisory in the general area of Fort Totten for PCBs [21].

6. Summary and conclusions

The default assumptions used for this risk assessment are reasonable maximum exposure factors. Thus, the assumptions are conservative of human health, and the calculated risks are likely greater than the actual risks. Another important thing to observe is that presently, all portions of Fort Totten are restricted areas. All the risk assessment analyses involve future use scenarios, if the property were transferred or sold for development as a park or waterfront housing area.

Presently, the risk to personnel who are employed at the installation is negligible. The risk to any present trespasser is also negligible. That is because these groups do not engage in sunbathing or swimming in Little Bay, or eat its fish, to any

appreciable extent. Therefore, for these groups, the exposure pathway is generally incomplete.

Using the RME values, it was determined that surface water and sediment ingestion presents little risk to adults or children. Ingestion of oysters and mussels caught in Little Bay is not harmful, either. Weekly, large portions of bottom dwelling fish, caught in Little Bay, could be a marginal health risk, to a child. But again, the most conservative assumption of mercury concentration in fish flesh, eaten weekly, over years, was used. Total surface water, body contact, by swimming, for example, is not a risk either, to adults or children, as shown in Section 5.3.1. While the concentration of mercury in fish does not pose a hazard, it is the largest contributor to the hazard index. The risk assessment for ingestion of fish is based upon the maximum concentration of mercury that was observed in 2 of 10 flounder samples. The concentration of mercury in the fish that is producing the risk, 0.27 mg/kg, is almost four times lower than the concentration of mercury that is the USFDA's limit for human consumption, 1 mg/kg [19]. USFDA suggests that you should eat only 7 oz (0.20 kg) of fish per week if it contains 1 mg/kg of mercury and 14 oz (0.40 kg) of fish with 0.5 mg/kg of mercury. Therefore, a receptor could consume 28 oz (0.79 kg) of the flounder a week with the highest level of mercury. If USFDA data or data from the ATSDR were used, the risk of eating fish would be significantly smaller and the hazard index for the bay would be less than 1. Therefore, the mercury in Little Bay does not pose a significant health risk when alternative, yet protective, health benchmarks are used.

Data collected by the U.S. National Oceanic and Atmospheric Administration [22] in surrounding areas of Long Island Sound, show mercury levels in the sediment up to 5 mg/kg. The concentrations of mercury in Little Bay were not substantially higher than the concentrations of mercury in sediment in other portions of Long Island Sound and New York Harbor; this suggests that the mercury in the Little Bay sediments is not the result of a release from Building 615. Therefore, clearing out and/or capping (i.e., with a fresh layer of sand), the sediment in the shoreline area of Little Bay would probably only realize a temporary benefit, since the surrounding sediments with the high mercury levels would eventually mix with the sediment that was cleared out by storms and currents.

In view of risks determined in exposure to mercury in the sediments in Little Bay, the optimal course of action is monitoring the sediments and surface water. Immediate action in terms of disturbing the sediments, is not recommended. Such action could potentially release mercury into the water column and increase the potential for movement into the surrounding biota.

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