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BASELINE ECOLOGICAL RISK ASSESSMENT

Newark Bay Study Area

Newark Bay, New Jersey

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ACRONYMS AND ABBREVIATIONS

2,3,7,8-TCDD	2,3,7,8-tetrachlorodibenzo-*p*-dioxin
ABS	Aquatic Biological Survey
ADD	average daily dose
AE	assessment endpoint
AhR	aryl hydrocarbon receptor
AIC	Akaike's Information Criteria
ANCOVA	analysis of covariance
ANOVA	analysis of variance
AOC	Administrative Order on Consent
ASMFC	Atlantic States Marine Fisheries Commission
BAF	bioaccumulation factor
Bay	Newark Bay
BAZ	biologically active zone
BERA	baseline ecological risk assessment
BHHRA	baseline human health risk assessment
B-IBI	Benthic Index of Biotic Integrity
BIC	benthic invertebrate communities
CBR	critical body residue
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act of 1980
CF	conversion factor
cfs	cubic feet per second
cm	centimeters
COC	constituent of concern
COPEC	constituents of potential ecological concern
CPG	Cooperating Parties Group
CSM	Conceptual Site Model
CV	coefficient of variation
cy	cubic yards
DDD	dichlorodiphenyldichloroethane

DDE	dichlorodiphenyldichloroethylene
DDT	dichlorodiphenyltrichloroethane
DF	dietary fraction
~~DO~~	~~dissolved oxygen~~
dw	dry weight
EFH	essential fish habitat
EPC	exposure point concentration
EPL	Experimental Pathology Laboratories, Inc.
ERL	effects range-low
ERM	effects range-median
FOD	frequency of detection
FS	feasibility study
FFS	Focused Feasibility Study
GSH	Glenn Springs Holdings, Inc.
HDP	Harbor Deepening Project
HQ	hazard quotient
HMW	high molecular weight
IR	ingestion rate
kg/kg bw-day	kilogram per kilogram body weight per day
LD50	lethal dose 50%
LMW	low molecular weight
LOAEC	lowest observed adverse effect concentration
LOAEL	lowest observed adverse effect level
LOE	line of evidence
LPR	Lower Passaic River
LPRSA	Lower Passaic River Study Area
ME	measurement endpoint
mg	milligrams
mg/kg	milligrams per kilogram
mg/kg bw/day	milligrams per kilogram body weight per day
~~mg/L~~	~~milligrams per liter~~

mm	millimeters
NAVD88	North American Vertical Datum 1988
NBSA	Newark Bay Study Area
ng/g	nanograms per gram
ng/kg-day	nanograms per kilogram per day
ng/L	nanogram per liter
NJDEP	New Jersey Department of Environmental Protection
NMDS	non-metric multidimensional scaling
NMFS	National Marine Fisheries Service
NOAEC	no observed adverse effects concentration
NOAEL	no observed adverse effect level
NOAA	National Oceanic and Atmospheric Administration
Normandeau	Normandeau Associates, Inc.
NY/NJ	New York/New Jersey
NYSDEC	New York State Department of Environmental Conservation
OCC	Occidental Chemical Corporation
OU	operable unit
PAH	polycyclic aromatic hydrocarbon
PC	principal component
PCA	principal component analysis
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-*p*-dioxin
PCDF	polychlorinated dibenzofuran
PE	polyethylene
POM	polyoxymethylene
ppth	parts per thousand
PRC	performance reference compound
PSU	practical salinity units
QAPP	Quality Assurance Project Plan
QA/QC	quality assurance/quality control
REMAP	Regional Environmental Monitoring and Assessment Program

RI	remedial investigation
RIWP	Remedial Investigation Work Plan
ROD	Record of Decision
SE3	Saline Estuarine Category 3
SLERA	screening level ecological risk assessment
SI	site investigation
SOC	Species of Concern
SQuiRT	Screening Quick Reference Table
SQT	sediment quality triad
SSD	species sensitivity distribution
SUF	site use factor
SumTU	sum of toxic units
SWQC	Surface Water Quality Criteria
TEF	toxic equivalency factor
TEQ	toxic equivalency
Tierra	Tierra Solutions, Inc.
TOC	total organic carbon
TRV	toxicity reference value
UCL	upper confidence limit
USACE	U.S. Army Corps of Engineers
USEPA	U.S. Environmental Protection Agency
USFWS	U.S. Fish and Wildlife Service
Windward	Windward Environmental LLC
WOE	weight of evidence
ww	wet weight
µg/kg	micrograms per kilogram
%	percent

~~1 INTRODUCTION~~

1 INTRODUCTION

The Diamond Alkali Superfund Site was added to the Superfund National Priorities List on September 21, 1984 because of contaminants present at the Site. Four different operable units (OUs) are associated with the Site (Figure 1-1): the former manufacturing plant and surrounding properties at 80 and 120 Lister Avenue (Operable Unit 1 [OU1]), the lower 8.3 miles of the Passaic River (Operable Unit 2 [OU2]), the Newark Bay Study Area (NBSA; Operable Unit 3 [OU3]), and the lower 17 miles of the Passaic River (Operable Unit 4 [OU4]; Lower Passaic River Study Area [LPRSA]; U.S. Environmental Protection Agency [USEPA] 2016).

Pursuant to the Administrative Order on Consent (AOC) under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA; Index 02-2004-2010; USEPA 2004), Glenn Springs Holdings, Inc., (GSH), on behalf of Occidental Chemical Corporation (OCC; the successor to Diamond Shamrock Chemicals Company [formerly known as Diamond Alkali Company]), has conducted a Baseline Ecological Risk Assessment (BERA) for the NBSA. The results and findings of this BERA will be incorporated into the Remedial Investigation (RI) for the NBSA, and evaluated with respect to potential remedial goals and options in the Feasibility Study (FS).

Baseline human health risk assessments (BHHRA) and ecological risk assessments (BERA) have been conducted for both the LPRSA and NBSA. These are on separate but parallel tracks, with coordination by the USEPA to ensure that the assessment approaches and structure are as comparable as possible, so that Diamond Alkali Superfund Site-wide risk management decisions can be made in a consistent manner.

This document presents the BERA for the NBSA. The objectives of this BERA are to evaluate potential exposure risks for aquatic organisms and wildlife (i.e., representative benthic invertebrates, fish, birds, mammals, and reptiles) that inhabit or utilize the Newark Bay environs, and provide a risk characterization that can be used to make informed risk management decisions for the NBSA.

1.1 Site Setting and Study Area Description

Newark Bay (Bay) is a 6.3-square-mile enclosed embayment on the western side of the New York/New Jersey (NY/NJ) Harbor Estuary. The Bay is adjacent to four large cities (Newark, Elizabeth, Bayonne, and Jersey City), and is fringed on its western side by port facilities, industrial facilities, and the Newark Liberty International Airport. On its northern side, the Hackensack and Passaic Rivers flow into the Bay, while on the southern side, the Bay is connected to NY Harbor (NY) and Raritan Bay (NJ) through two tidal straits: the Kill van Kull and Arthur Kill, respectively. The NBSA has been defined as the Bay and portions of key tributaries, including the Hackensack River, Arthur Kill, and Kill van Kull. The Bayonne and Goethals Bridges are not the OU3 boundaries themselves (Figure 1-2). The Passaic River is not included in the definition of the NBSA, as it is presently being investigated as a separate OU (OU4). However, investigations of the Passaic River and NBSA OUs are being conducted in a comparable manner and

with careful consideration of their linkages for the purposes of CERCLA management decision-making and broader environmental management considerations.

The Bay is a partially mixed estuary, meaning that there is some density stratification of saline and freshwater (Suszkowski 1978; U.S. Army Corps of Engineers [USACE] 2007b). Salinity in the Bay has been reported to range from approximately 14 to 24 parts per thousand (ppth), with an average annual salinity of approximately 20 ppth (USACE 2007b). Salinity measured in Newark Bay as part of the Physical Water Column Monitoring program between April and July 2010 ranged from 3.7 to 24.7 practical salinity units (PSU) with an average of 18.6 PSU[1] (AECOM 2019). Estuaries can have a wide range in salinity from freshwater to marine environments (0.5 to 30 ppth; USACE 2007b). The tides in the Bay area are semi-diurnal, with a mean tide range of 5.1 feet and a spring tide range of 6.1 feet. The mean tide level is 5.9 feet above North American Vertical Datum of 1988 (USACE 1997). Tidal currents are the primary mechanism of water flow within the Bay and are strongest (both ebb and flood) in the Arthur Kill and Kill van Kull. The dominant freshwater contributions to the Bay are from the Passaic and Hackensack Rivers at the northern end of the Bay, while other freshwater sources to the Bay include smaller tributaries like the Elizabeth River, Peripheral Ditch, and Piersons Creek, among others (Figure 1-1). Annual flow computations by Blumberg et al. (1999) show that approximately 60 percent (%) of annual flows into the Bay come from the Kill van Kull and 34% come from the Hackensack and Passaic Rivers combined, while net discharge to the Raritan Bay occurs through the Arthur Kill. Salinity is highest and density stratification is weakest in the Kill van Kull due to high volume exchange with NY Harbor (Suszkowski 1978).

The Newark Basin, in which the NBSA is located, is underlain by sedimentary rocks (sandstones, shales, limy shales, and conglomerates), igneous rocks (basalt and diabase), and metamorphic rocks (schists and gneiss). Sediments found on the bottom or just below the bottom of the Bay are mostly fine-grained within the range of silt and clay-sized particles (USACE 1997), which is consistent with the grain size analysis from Phase I, II (Tierra 2014c) and III (GSH ~~2019~~2020) however, some parts of the Bay consist of 50% to 75% fine sands (GSH ~~2019~~2020). The pattern of sediment types (sand/gravels versus silt/clays) is indicative of fluvial sediment input at the northern end of the Bay and tidal exchange sedimentation at the southern end.

Hydrodynamics within the Bay are influenced primarily by four inter-related physical forces: 1) tributary flows from the Passaic and Hackensack Rivers; 2) astronomical forcing (tides); 3) local and regional meteorological events (wind and storm effects); and 4) gravitational forcing (density-driven circulation) (Chant 2006; Herrington et al. 2002; Wakeman 2006). These primary influences are responsible for the variations in hydrodynamic and sediment transport characteristics within the system (Herrington et al. 2002).

A description of the environmental history of the Newark Bay region, and detailed characterization of the Bay ecosystem are provided in Section 2.

[1] PSUs are a more modern way to express salinity based on electrical conductivity, but are nearly identical to older salinity measures in ppth.

1.2 Superfund Site History and Participation

The NBSA is part of the Diamond Alkali Superfund Site, which was added to the Superfund National Priorities List on September 21, 1984 because of 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD) present at the site. The USEPA's response began at the former facility located at 80 and 120 Lister Avenue in Newark, New Jersey (OU1). 2,3,7,8-TCDD was a by-product of one of the manufacturing processes (USEPA 2016a).

Four different OUs are associated with the Diamond Alkali Superfund Site today (Figure 1-1):

- OU1: 80 and 120 Lister Avenue
- OU2: Lower 8.3 miles of the Passaic River
- OU3: NBSA
- OU4: Lower 17 miles of the Passaic River

In 1990, a Consent Decree was entered into by the USEPA/State of New Jersey and OCC to implement and maintain an interim remedy, documented in a 1987 Record of Decision (ROD), at 80 and 120 Lister Avenue for OU1 (United States District Court for the District of New Jersey 1990). The primary contaminants of concern identified in the ROD for OU2 are polychlorinated dibenzo-p-dioxins and dibenzofurans (dioxins and furans, or PCDD and PCDFs), polychlorinated biphenyls (PCBs), mercury, dichlorodiphenyltrichloroethane and its primary breakdown products, dichlorodiphenyldichloroethane and dichlorodiphenyldichloroethylene (referred to as total DDx), copper, dieldrin, and polycyclic aromatic hydrocarbons (PAHs).

In 2007, an AOC was signed by the USEPA and the Cooperating Parties Group (CPG; USEPA 2007a) to conduct an RI/FS) for OU4 (and OU2). The primary contaminants of concern identified for the Diamond Alkali Superfund Site in the 2007 AOC include PCDDs, PCDFs, PCBs, PAHs, pesticides, and inorganics/metals.

In 2004, the USEPA and OCC signed an AOC under CERCLA (Index 02-2004-2010; USEPA 2004) in which OCC agreed to conduct an RI/FS for the NBSA (USEPA 2004). This work was formerly implemented, through April 2017, by Tierra Solutions, Inc. (Tierra) on behalf of OCC. GSH is currently performing the work on OCC's behalf.

Cleanup activities performed or planned to date at the Diamond Alkali Superfund Site include the following:

- On September 30, 1987, the ROD for OU1 was finalized. The USEPA-selected interim remedy, which was completed in 2001, included constructing slurry and flood walls, capping the properties, and pumping and treating of groundwater to reduce its migration.
- In June 2008, the USEPA and OCC signed an agreement to remove 200,000 cubic yards (cy) of contaminated sediment from the river adjacent to OU1. Phase I was completed in 2012 and included dredging, dewatering, and offsite disposal of 40,000 cy of contaminated sediment. Phase II was

intended to include removal of the remaining 160,000 cy with disposal in a confined disposal facility. Remediation of the Phase II area is being integrated with the OU2 remedy.

- In June 2012, the USEPA and CPG signed an agreement to address contaminants found at the surface of a mudflat on the east bank of the Passaic River at river mile 10.9 in Lyndhurst, New Jersey. In 2013 and 2014, surface sediment was dredged and a cap was constructed.
- On March 3, 2016, the ROD for OU2 was finalized. The selected remedy consists of capping with sediment dredging to accommodate cap construction. The design of the OU2 remedy is underway (USEPA 2018).

The NBSA, which is the subject of this BERA, is presently in the RI/FS process. In addition to this BERA, an RI and BHHRA are being conducted for the NBSA, and the FS will follow.

1.3 Remedial Investigation Objectives

The RIs in the NBSA, designed and conducted according to the requirements specified in the RI/FS AOC and associated statement of work, were developed to generate additional data that will be used to produce the RI Report, an FS Report, and ultimately support the development of a ROD for the NBSA. The NBSA AOC defined three RI-related goals:

1. Nature and Extent of Sediment Contamination: Determine the horizontal and vertical distribution and concentrations of PCDDs, PCDFs, PCBs, PAHs, pesticides, and metals for NBSA sediments.
2. Risk Assessment: Determine the primary human and ecological receptors (endpoints) of PCDD-, PCDF-, PCB-, PAH-, pesticide-, and metals-contaminated sediments in the NBSA.
3. Source Identification: Determine the significant direct and indirect continuing sources of PCDDs, PCDFs, PCBs, PAHs, pesticides, and metals to the sediments in the NBSA (USEPA 2004).

As described in the RI/FS Phase I Investigation Work Plan (Phase I RIWP; Tierra 2005) and Phase II Remedial Investigation Work Plan (Phase II RIWP; Tierra 2007), sediment and related investigation activities were carried out in two initial phases. On September 9, 2015, the USEPA submitted a memorandum to Tierra that described a need to characterize subunits of the Bay at an appropriate scale for RI/FS decision-making because the existing data only allowed for Bay-wide decision making (USEPA 2015a). A Phase III surface sediment investigation was designed and implemented in 2016 Newark Bay Study Area Phase III Sediment Investigation Quality Assurance Project Plan Amendment (Phase III QAPP; Tierra 2016). Sampling was conducted in a systematic, stratified manner to collect sufficient data to characterize and statistically compare surface sediment data in the various geomorphic zones and defined subunits of the Bay, and a number of previous Phase I/II locations were resampled to provide data for recent (i.e., previous decade) trend analyses. This program was also designed/intended to provide a baseline surface sediment sampling grid for the NBSA to utilize for future sampling events to conduct long-term trend analyses.

In addition to the RI, a number of focused investigations were conducted to provide the necessary data and information to conduct a BHHRA and BERA for the NBSA. These include the development of the Screening-Level Ecological Risk Assessment (SLERA) Report (Attachment 1; Battelle 2008), the Problem Formulation Report (Attachment 2; Tierra 2013a), the Newark Bay Study Area Reconnaissance Survey

Report (Attachment 3; Tierra 2015a), sediment quality triad (SQT) and porewater sampling and analyses (GSH 2017a), crab and clam sampling and analyses (GSH 2017b), and fish sampling and analyses (Tierra 2017). All work for this BERA was carried out in accordance with USEPA-approved sampling and quality assurance project plans (Tierra 2014a, 2014b, 2015b). Section 4 describes the various investigations and their application to this BERA.

1.4 Report Organization

Together with this introductory section, this BERA is organized as follows:

- Section 2 – Ecosystem Characterization
- Section 3 – Problem Formulation
- Section 4 – Data Results and Evaluation
- Section 5 – Surface Water Assessment
- Section 6 – Benthic Invertebrate Assessment
- Section 7 – Fish Assessment
- Section 8 – Bird Assessment
- Section 9 – Mammal Assessment
- Section 10 – Diamondback Terrapin (Reptile) Assessment
- Section 11 – Risk Summary and Conclusions
- Section 12 – References.

2 ECOSYSTEM CHARACTERIZATION

The following is a summary of the present NBSA environmental setting, and the history of changes to the Bay that have created the present ecosystem under investigation in the RI/FS and this BERA. Detailed descriptions of the NBSA system are provided in the RI Report presently being developed (GSH ~~2019, in preparation~~2020).

2.1 Environmental Setting

The Bay is central to one of the most urbanized and industrialized areas in the United States. It has experienced more than two centuries of environmental degradation attributable to many factors, including shoreline and land development (USACE 2006a), wetlands/habitat loss, garbage and sewage disposal, dredging and dredged material disposal, and releases of contaminants from a variety of sources and locations (Iannuzzi et al. 2002).

2.1.1 History

The environmental history of Newark Bay parallels the development of the New York City metropolitan area. Most shipping and economic development in the 19th century clustered around Manhattan and Brooklyn, but as the pace of development quickened in the first half of the 20th century, the Bay eventually supplanted Manhattan as the primary port by the middle of the 20th century. Over that period of development, approximately 80 to 90% of the pre-existing shoreline of the Bay was developed and ecological habitats correspondingly diminished (Iannuzzi et al. 2002; USACE 2007a, 2009a).

Newark and Elizabeth (known in its early history as Elizabethtown and Elizabethport), much like surrounding communities, started as small towns surrounded by farms, forest, and wetlands (Hassler 1844). By the 1840s, railways bisected wetlands and connected Newark to Jersey City and Elizabeth, setting the pre-conditions for future industrial development. Newark was home to several chemical companies producing various chemical products and raw materials by 1850 (Cunningham 1954; Zdepski 1992). By 1900, Newark was the largest industrial-based city in the United States, with well-established industries, including petroleum refining; shipping; tanneries; creosote wood preservers; metal recyclers; and manufacturers of materials, including rubber, rope, textiles, paints and dyes, pharmaceuticals, raw chemicals, leather, and paper products (Meyers 1945; Cunningham 1954, 1966; Brydon 1974; Halle 1984; MacRae's Blue Book, Inc. 1986; Galishoff 1988). By the early 20th century, housing developments covered most of the Bayonne peninsula, and the cities of Newark and Elizabeth on the west side were greatly expanded as the growing industrial manufacturing complex in the Newark region replaced agriculture and large numbers of people moved to the area. The Bay continued to evolve into what is now a highly modified, major shipping port.

The development of the Bay port system required extensive land development, achieved through "reclamation" of the meadowlands (wetlands) along the Bay and the Hackensack River during the 20th century. As the area's population and industrial development grew, transportation needs increased, and a large network of roads, bridges, airports, and port facilities was constructed. Today, Newark—the largest city in New Jersey—is a trade and transportation center. One of Newark's important functions as a

transportation center is the transfer of goods from cargo vessels in the Bay ports to the many railroad and truck lines that serve the region and beyond. Land use along the northern, western, and southern shoreline of the Bay is high-density commercial and industrial/commercial development. A highly developed network of highways, combined sewer overflows, storm sewer outfalls, and publicly owned treatment works also exist within the Bay and its watershed (Mueller et al.1982).

Industrialization of the Bay was facilitated by development of navigational channels in the NY/NJ Harbor Estuary starting in the mid-1800s through channel deepening and widening by the USACE New York District (see Figure 1-2 for current navigation channel system). Table 2-1 presents the Bay's history of dredging, navigation channel widening, and wetlands alteration. Bathymetry surveys have been conducted since the early 1900s by the National Oceanic and Atmospheric Administration (NOAA) and USACE, and historical nautical charts prepared by the NOAA illustrate the shaping of the Bay.

The depth and width of the navigational channels have grown continually since the early 1900s (USACE 2006a), although channels in some adjacent water bodies (e.g., the Lower Passaic River [LPR] and Hackensack River) are no longer maintained to their previously authorized depths. The navigation channel in the LPR above river mile 1.7 (i.e., above the Route 1-9 Lincoln Highway Bridge shown in Figure 1-2) has been deauthorized. The navigation channel in the Bay north of the port channels is not maintained to previously authorized depths, even though it is still subject to large vessel traffic (e.g., wastewater treatment sludge barges). Substantial maintenance dredging of the existing channels is required. During the most recent maintenance dredging (2016), approximately 285,000 cy of material were removed from critically shoaled areas of the Port Newark approach channels. Currently, channel conditions are being monitored.

2.1.2 Land Use and Shoreline Characterization

A present land use map of the NBSA is presented on Figure 2-1. The NBSA shoreline is primarily comprised of the following types:

1. Bulkhead – Extent of shoreline where majority of vertical face consists of wood timbers, metal sheet pile, or large stone/cement blocks; ~40% of the NBSA shoreline is classified as bulkheads.
2. Mixed Intertidal – Extent of shoreline where a combination of riprap, sand, and/or emergent vegetation is present. Mixed ~~industrial~~intertidal shorelines comprise ~10% of the NBSA shoreline.
3. Riprap – Extent of shoreline dominated by sloped banks that are covered by cobble to boulder sized stones and/or concrete rubble. Riprap covers ~~approximately~~ ~30% of the NBSA shoreline.
4. Vegetation – Extent of shoreline with greater than 50% emergent vegetation~~. Approximately~~ ; ~20% of the NBSA shoreline is classified as vegetated.

Human use of the NBSA upland areas is primarily industrial and commercial. Recreational use is limited due to access limitations of the shoreline types and surrounding land use. Based on a reconnaissance survey (Tierra 2015a), five human use categories were established and described as follows along with the percentage of upland characterization adjacent to the shoreline where each was observed:

- Disturbed Uplands – open space areas that have been abandoned and overgrown with terrestrial vegetation (e.g., trees, scrub/shrub) - ~~18~~~20%

- Undisturbed Uplands – areas consisting of natural vegetation that provides habitat for wildlife species (natural vegetation does not include landscaped vegetation which is part of the Residential category described below) – ~~26~~~25%
- Industrial/Commercial – areas consisting of industrial/commercial use, such as, but not limited to, manufacturing facilities, ports, and shipyards – ~~36~~~35%
- Recreational – areas where recreational activities are dominant (e.g., parks, fishing spots, walking trails) – ~~12~~~10%
- Residential – areas consisting of, but not limited to, residential homes, apartment complexes, and schools – ~~11~~~10%.

2.1.3 Contamination

Numerous former and current industrial and manufacturing facilities in the Newark metropolitan area serve(d) as potential point and non-point source discharges to the sediment environment. These industries include (but are not limited to) metals refining, dye manufacturing, tanning, soap and candle making, lumber processing, hat manufacturing, carriage building, shoe making, petroleum processing, chemical manufacturing, pesticide and herbicide production, paper and textile manufacturing, copper rolling, wire manufacturing, coke making, and manufactured gas plants (Tierra 2013a; Iannuzzi et al. 2002). Industrial dischargers located along the waterfront and inland areas of the Passaic River, Hackensack River, Arthur Kill, and Newark Bay have contributed to sediment contamination in the NBSA.

The NBSA is known to be contaminated with a wide variety of organic compounds and inorganic chemicals (i.e., PCDD/PCDFs, PCBs, PAHs, pesticides, herbicides, semivolatile organic compounds, volatile organic compounds, inorganics/metals, and other organic compounds). There are many known sources of contaminants to the Bay, including:

- Industrial discharges
- Publicly owned treatment works, combined sewer overflows, storm sewers, and other non-point sources
- Spills, leaks, and accidental discharges from marine and industrial sources
- Atmospheric deposition and groundwater discharges
- Tributary inputs from each of the sources listed, and transport of re-mobilized legacy sediments from tributaries.

Existing contamination in the NBSA is primarily from historical and present sources from each of these categories, which in combination have been released over more than a century paralleling the urban and industrial history of the Bay. The relative influence or importance of these various sources is not easily quantifiable, and likely varies depending on the geographic area, contaminant group, temporal fate and transport processes, and the depth of the contaminated sediment layer under consideration. Additional

information regarding sources of contaminants in the Bay is provided in the RI Report (GSH ~~2019, in preparation~~2020) and *Report on Investigation of Sources of Pollutants and Contaminants* (Tierra 2006[2]).

Sediment-based risk in this BERA is evaluated using sediment chemistry data from surface sediment samples (0 to 6 inches below sediment surface). In some areas of the Bay, higher concentrations of contaminants are found in deeper sediments, particularly in areas of the subtidal flats that have been historically disturbed (i.e., former channels or structures). However, evidence presented in the RI (GSH ~~2019~~2020), indicate that there are no expected changes to the ecosystem in the foreseeable future that would alter the exposure conditions for the species evaluated in this BERA. Evaluation of long-term trends using data from deep sediment cores indicates that contaminant concentrations have been declining. Temporal trends evaluated over the period of RI sediment sampling (i.e., 2005-2016) indicate that a declining trend continues (GSH ~~2019~~2020). Therefore, the potential for exposure of receptors to deeper sediments are not evaluated in this BERA.

2.1.4 Habitats

The following is a characterization of ecological habitats in the NBSA. Data and information for this characterization are integrated from the Problem Formulation (Attachment 2; Tierra 2013a) and Reconnaissance Survey (Attachment 3; Tierra 2015a) reports. Collectively, the data and information compiled to date under the RI process provides a good understanding of the Bay ecosystem.

Wetland and tributary losses, and present-day shorelines covered by bulkheads, riprap, buildings, and impervious surfaces (i.e., pavement and concrete) limit the available wildlife habitat in the NBSA, particularly nesting and foraging areas for birds and mammals (Iannuzzi et al. 2002). Table 2-2 provides the acreages of the geomorphic areas of the Bay. Subtidal flats, including the portions that were historically disturbed, are the primary habitat areas for aquatic organisms in the Bay. Intertidal areas (Table 2-2) and vegetated wetlands (Figure 2-1) comprise only a small portion of the overall shoreline area. Degraded structures and pilings offer some habitat for aquatic species. Shooters Island offers some forested and wetland habitat for birds. North and south of the NBSA, there are extensive tidal wetlands associated with the Hackensack River and the Arthur Kill that provide wildlife habitat for birds and mammals. Despite the changing habitat conditions in the ecosystem during the 20th century, the Bay acts as a spawning ground, migratory pathway, and a nursery/foraging area for a variety of aquatic organisms.

Table 2-2. Area Coverage in the Newark Bay Study Area

Geomorphic Area	Area (acres)	Percent of NBSA Area	Average Depth[a] (feet NAVD88)
Subtidal Flats	1737.9	42.6	7.5
Historically Disturbed Subtidal Flats	251.4	6.2	8.4
Transitional Slopes	411.1	10.0	26
Navigation Channels/Port Channels	1277.3	31.3	50[b]
Intertidal Areas	28.9	0.7	3.2

[2] This document has not been approved by the USEPA.

Geomorphic Area	Area (acres)	Percent of NBSA Area	Average Depth[a] (feet NAVD88)
Industrial Waterfront Areas	346.4	8.5	17
Confined Disposal Facility[c]	27.4	0.7	6.1

Notes:
[a] Average depths were tide-corrected from the 2012/2013 bathymetric survey (Tierra 2013b).
[b] Average depth of the Navigation Channels/Port Channels is estimated based on the Harbor Deepening Project completed in 2016.
[c] The acreage of the confined disposal facility is shown separately, however it is part of the subtidal flats geomorphic area.
NAVD88 = North American Vertical Datum of 1988

2.1.4.1 Terrestrial (Shoreline) and Intertidal Wetland Habitats

Upland vegetation along the NBSA generally consisted of a patchy mixture of early successional and ruderal plant species, indicative of the high level of shoreline disturbance. Dominant canopy species included eastern cottonwood (*Populus deltoides*), locust trees (*Gleditsia* spp. or *Robinia* spp.), and catalpa (*Catalpa speciosa*) with an understory of honeysuckle (*Lonicera* spp.) and sumac (*Rhus* spp.). A total of 22 common plant and tree species were identified during the reconnaissance survey (Table 2-3). While this list is not comprehensive, it provides a good representation of the types of plant and tree species observed along the shoreline.

Table 2-3. Vegetation Species Observed During the 2013 Reconnaissance

Common Name	Scientific Name
Aspen	*Populus* spp.
Birch	*Betula* spp.
Black locust	*Robinia pseudoacacia*
Boneset (unidentified)	*Eupatorium* spp.
Catalpa	*Catalpa speciosa*
Chokecherry	*Prunus virginiana*
Common reed	*Phragmites australis*
Eastern cottonwood	*Populus deltoides*
Goldenrod (unidentified)	*Solidago* spp.
Groundsel tree	*Baccharis halimifolia*
Honeysuckle	*Lonicera* spp.
Jewelweed (unidentified)	*Impatiens* spp.
Locust (unidentified)	*Gleditsia* spp. or *Robinia* spp.
Pokeweed	*Phytolacca americana*
Princess trees	*Paulownia tomentosa*

Common Name	Scientific Name
Ragwort	*Jacobaea vulgaris, syn. Senecio jacobaea*
Red maple	*Acer rubrum*
Smooth cordgrass	*Spartina alterniflora*
Sumac	*Rhus* spp.
Sunflower	*Helianthus annuus*
Sycamore	*Platanus occidentalis*
Willow	*Salix* spp.

As is characteristic of disturbed upland habitats, species diversity was low and dominated by fast-growing invasive and/or short-lived vegetation. These upland habitats were small in size, lacked a consistent canopy and understory, and were surrounded by a mixture of commercial, industrial, and residential areas. Due to the limited size and complexity of these habitats, upland areas adjacent to the NBSA do not provide much in the way of suitable habitat for ecological receptors beyond cover, resting, or limited foraging.

The intertidal wetlands (Intertidal Areas) of the NBSA primarily consist of emergent common reed (*Phragmites australis*) and saltmarsh cordgrass (*Spartina alterniflora*). In general, the cordgrass occupies the more waterward areas of the intertidal zone, while common reed is present in the more landward areas of the intertidal zone and transition into the upland shoreline areas.

Shooters Island, located in the southern portion of Newark Bay, is one of the major forested areas of Newark Bay. It is partially wooded with species such as black locust (*Robinia pseudoacacia*), tree-of-heaven (*Ailanthus altissima*), and Japanese honeysuckle (*Lonicera japonica*). Small patches of saltmarsh containing a mix of common reed and cordgrass occur around the island's shoreline, intermixed with scattered debris, rotting docks, abandoned buildings, shipwrecks, and barges (U.S. Fish and Wildlife Service [USFWS] 1997). Despite its impacted condition, Shooters Island continues to serve as a bird sanctuary and refuge for migratory birds such as gulls, cormorants, and osprey, which have recently been observed on the island (Bernick and Craig 2008).

Additional forested areas occur along the eastern tip of Kearny Point, just south of the Elizabeth Marine Terminal, along the eastern side of Bayonne Park, and in Richard A. Rutkowski Park. Sporadic forested areas also occur between Veterans and City Parks, in Mariners Marsh, and around the Newark Bay Bridge (Figure 2-1).

Barren lands, consisting of recently cleared or disturbed areas where soils have been disturbed (e.g., coal cinders are often a major soil constituent), are common around the Bay shorelines. Vegetation consists of tolerant, often introduced, species. Vegetation is present in scattered patches, offering little cover for wildlife, particularly in winter. Garbage and debris items are present in the vacant lots/fields. Typical plant species include goldenrod (*Solidago canadensis* or *Solidago virgaurea*), mugwort (*Artemisia vulgaris*), raspberry (*Rubus* spp.), Asian bittersweet (*Celastrus orbiculatus*), pokeweed (*Phytolacca americana*), and saplings of royal paulowina (*Paulownia tomentosa*), mulberry (*Morus* spp.), tree-of-heaven, and black

locust. Typical wildlife observed includes terrestrial, urban-tolerant species (United States Coast Guard 2010).

Despite the highly developed shoreline, the NBSA provides habitat for some wildlife, primarily birds. A total of 34 different species of wildlife were observed during the 2013 NBSA reconnaissance survey (Table 2-4; Tierra 2015a). While the majority of observations were birds, several mammals (primarily urban opportunistic species, such as rats and coyotes) were observed throughout the NBSA. Also, one turtle was observed along an area of emergent vegetation. More detailed discussions of aquatic organisms and wildlife that inhabit or utilize the NBSA are presented in Sections 2.3 through 2.8.

Table 2-4. Wildlife Species Observed During the 2013 Reconnaissance

Common Name	Scientific Name
Birds	
American crow	*Corvus brachyrhynchos*
Belted kingfisher	*Megaceryle alcyon*
Black duck	*Anas rubripes*
Blue-winged teal	*Anas discors*
Brant	*Branta bernicla*
Canada goose	*Branta canadensis*
Grey Catbird	*Dumetella carolinensis*
Common sandpiper	*Actitis hypoleucos*
Common tern	*Sterna hirundo*
Double-crested cormorant	*Phalacrocorax auritus*
Great blue heron	*Ardea herodias*
Herring gull	*Larus argentatus*
House wren	*Troglodytes aedon*
Killdeer	*Charadrius vociferus*
Laughing gull	*Leucophaeus atricilla*
Little green heron	*Butorides virescens*
Mallard	*Anas platyrhynchos*
Northern mockingbird	*Mimus polyglottos*
Mourning dove	*Zenaida macroura*
Osprey	*Pandion haliaetus*
Pied-billed grebe	*Podilymbus podiceps*
Red-winged blackbird	*Agelaius phoeniceus*

Common Name	Scientific Name
Ring-billed gull	*Larus delawarensis*
White egret	*Ardea alba*
Mammals	
Cottontail rabbit	*Lepus sylvaticus*
Domestic cat	*Felis catus*
Domestic dog	*Canis lupus familiaris*
Eastern coyote	*Canis latrans*
Muskrat	*Ondatra zibethicus*
Norway rat	*Rattus norvegicus*
Raccoon	*Procyon lotor*
Striped skunk	*Mephitis*
White-tailed deer	*Odocoileus virginianus*
Reptile	
Diamondback terrapin	*Malaclemys terrapin*

2.1.4.2 Shallow Subtidal and Deepwater Habitats

Nearly half of the NBSA bottom area is composed of shallow subtidal flats (< 8 feet average depth), and an additional 40% is maintained navigation channels and their transitional side slopes (Table 2-2). ~~The~~These maintained navigation channels are regularly dredged deep water areas between approximately 40 and 50 feet in depth, and contain recently deposited sediments that are likely derived from sloughing of sediments from their transitional slopes and, to a lesser extent, general sediment transport processes in the water column (GSH ~~2019, in preparation~~2020).

The subtidal flats are the most stable and important aquatic habitat in the NBSA, providing long-term stability for established benthic invertebrate communities, and important foraging areas for aquatic (fish, shellfish) and semi-aquatic (diving birds and mammals) organisms. The maintained deep water navigation channels are ~~a~~an important but more transient habitat type, providing refuge and foraging areas for various aquatic organisms, ~~but a~~ including preferential foraging for some fish such as sturgeon and flounder. These are a less stable environment ~~that~~than other habitats in the NBSA, as the sediments are dredged on a fairly regular basis by the USACE to maintain and deepen the navigation capabilities to the Ports within the Bay. The regular dredging precludes the long-term establishment and maintenance of ~~long-term~~ stable (i.e., Stage III fully developed and undisturbed) benthic invertebrate (i.e., infaunal) communities due to the periodic dredging operations (Table 2-1). The regular removal of sediments by dredging also influences the contaminant exposure potential in these channels, as discussed in Section 6.

Despite the changing habitat conditions in the ecosystem during the 20th century, the Bay acts as a spawning ground, migratory pathway, and a nursery/foraging area for a variety of aquatic organisms.

Essential fish habitats were identified for 19 finfish species in the Bay (USACE 2013a; NMFS 2019) are presented by life stage in Table 2-5. The identified fish species have habitats designated in the Bay for one or more important life stages (i.e., eggs, larvae, juveniles, adults, or spawning adults). The shortnose and Atlantic sturgeon (*Acipenser brevirostrum* and *A. oxyrhynchus*) are federally listed as protected species in the NBSA (USACE 2013a).

Table 2-5. Essential Fish Habitat Designations by Life Stage for Newark Bay

Species	Eggs	Larvae	Juveniles	Adults	Spawning Adults
Summer flounder (*Paralichthys dentatus*)	–	✓	✓	✓	–
Winter flounder (*Pseudopluronectes americanus*)	✓	✓	✓	✓	✓
Windowpane flounder (*Scopthalmus aquosus*)	✓	✓	✓	✓	✓
Atlantic mackerel (*Scomber scombrus*)	–	–	✓	✓	–
King mackerel (*Scomberomorus cavalla*)	✓	✓	✓	✓	–
Spanish mackerel (*Scomberomorus maculates*)	✓	✓	✓	✓	–
Dusky shark (*Carcharhinus obscurus*)	–	✓	✓	–	–
Sand tiger shark (*Carcharias taurus*)	–	✓	✓	–	–
Sandbar shark (*Carcharhinus plumbeus*)	–	✓	✓	–	–
Clearnose skate (*Raja eglanteria*)	–	–	✓	✓	–
Little skate (*Leucoraja erinacea*)	–	–	✓	✓	–
Winter skate (*Leucoraja ocellata*)	–	–	✓	✓	–
Atlantic butterfish (*Peprilus triacanthus*)	–	✓	✓	✓	–
Atlantic sea herring (*Clupea harengus*)	–	✓	✓	✓	–
Black sea bass (*Centropristis striata*)	–	–	✓	✓	–
Bluefish (*Pomatomus saltatrix*)	✓	✓	✓	✓	–
Cobia (*Rachycentron canadum*)	✓	✓	✓	✓	–
Red hake (*Urophycis chuss*)	✓	✓	✓	✓	–
Scup (*Stenotomus chrysops*)	✓	✓	✓	✓	–

Notes:
✓ indicates habitat is suitable for life stage.
– indicates habitat is not suitable for life stage.

Source: (USACE 2013a, National Marine Fisheries Service [NMFS] 2019)

2.2 Plankton

There have been limited studies on the planktonic organisms present in the NBSA. The available data suggest that the Bay supports a variety of phytoplankton and algal communities that are typical of the NY/NJ Harbor Estuary. The plankton are periodically surveyed by the New Jersey Department of Environmental Protection (NJDEP 2009). In 2005, as in previous years, the plankton communities were dominated by a diverse assemblage of diatoms in mild to full bloom proportions. Dinoflagellates detected during this season were *Gyrodinium undulans*, *Olisthodiscus luteus*, and *Prorocentrum micans*. In addition, *Pseudo-nitzschia* spp. and *Dinophysis* spp., both potentially toxic species, were detected below bloom or toxic concentrations (NJDEP 2009).

The zooplankton community of the NBSA can be divided into two major categories: permanent holoplankton (which include various forms of small, sometimes microscopic organisms, such as protozoans and copepods) and temporary meroplankton (which include larval stages of shallow-water invertebrates and fish). Among the meroplankton are the ichthyoplankton, consisting of egg, larval, and juvenile stages of fish. A discussion of the ichthyoplankton surveys that have been performed in the NBSA are presented in Section 2.4 on fish. Potential risks to planktonic communities in the NBSA are evaluated in Section 5.

2.3 Benthic Invertebrates

A number of historical studies have been conducted on the benthic invertebrate communities of the NBSA (Tierra 2004). These studies typically demonstrated that the benthic invertebrate communities throughout the Bay were characterized by relatively low diversity and abundance, and that the benthic environment was stressed from various pollutants and anoxic conditions. Species lists in the historical studies were dominated by polychaete worms, oligochaete worms, and small bivalves (clams/mussels). These organisms serve as a forage base for a variety of crustaceans, fish, and wading birds. A characterization of the infaunal benthic invertebrate communities in the NBSA, synthesized from these various studies, is provided in Section 2.3.1. A discussion of the shellfish communities (i.e., mollusks and crustaceans) in the Bay is provided in Section 2.3.2.

2.3.1 Benthic Invertebrate Community

A survey of benthic habitats in the NY/NJ Harbor Estuary conducted in June and October 1995 (Iocco et al. 2000) indicated that Newark Bay bottom sediments were predominantly characterized as silt with occasional sand. Grab samples from approximately one-half the stations sampled contained benthic infauna, consisting of predominantly polychaetes (e.g., *Streblospio benedicti*) and bivalves (e.g., *Mulinia lateralis* and *Mya arenaria*). Benthic habitats, such as large clam beds and mats of amphipods (*Ampelisca abdita*) that were seen elsewhere in the NY/NJ Harbor Estuary, were not observed in the Bay. Sediment gas content at several stations in the Bay indicated high pollutant or organic content. Surveys conducted by NOAA in 1993 and 1994 and USACE in 1995 and 1996 also found the Bay to be dominated by polychaetes and small clams, with a low diversity and abundance of pollution-indicative species (NOAA 1994b; USACE 1997). Benthic infaunal abundance and species composition were also found to increase in the late winter and early spring months and decline in the summer (USACE 1997). A table of dominant species from the various studies conducted to date in Newark Bay are presented in Table 2-6.

A study conducted in October 2005 at 14 stations in Newark Bay utilized sediment profile imagery and supplemental grab samples to determine the biologically active zone (BAZ; Tierra 2008b). Results of the study indicate that the surface sediment is structured by physical (e.g., currents and sediment movement) and biological processes (e.g., movement of infaunal organisms), both of which can affect the bioavailability of chemical constituents. The fauna and biogenic structures observed were sufficient to bioturbate sediment to the estimated total BAZ depths of 13.7 to 16.4 centimeters (cm; average of approximately 6 inches or 0.5 feet). The deepest vertically burrowing species observed were large individuals of *Macoma balthica* and *M. arenaria* clams. The species with the potential to create the deepest convoluted burrow galleries were *Glycera* spp. and *Nereis* spp. Other species found in the grab samples were the burrowing isopod *Cyathura polita* and large-bodied, tube-building polychaetes (*Diopatra cuprea*), maldanids (*Pectinaria gouldii*), and the amphipod Ampelisca spp. Most of the stations sampled in this investigation were composed of benthic assemblages in a Stage III (relatively well-developed) successional stage that suggests stability in the sediment structure (Tierra 2008b).

In 1993 to 1994, 1998 to 1999, 2003, 2008, and 2013, USEPA (under the Regional Environmental Monitoring and Assessment Program [REMAP]) collected 140 randomly located benthic invertebrate samples in the area referred to as the Newark Bay sub-basin, which, in addition to the Bay, included stations in the Arthur Kill, Lower Passaic River, and Hackensack River (Adams 2016; USEPA 2003a) (Figure 2-2). A total of 28 locations were sampled within the Newark Bay sub-basin each year of the study, and between 11 and 18 stations (sampling year dependent) were within the NBSA. A summary of the benthic community indices calculated under the REMAP by year are presented in Table 2-7 (NBSA stations only). The pre-NBSA RI period (i.e., prior to 2004) REMAP studies concluded that the overall benthic invertebrate community was characterized by low abundance and diversity and dominated by pollution-indicative species. The 1993 to 2003 Benthic Index of Biotic Integrity (B-IBI) values, an index calculated based on various structure and abundance metrics, indicated that as much as 90% of Newark Bay was classified as moderately to highly impacted (Figure 2-3). In the 1998 study, all but one station within the NBSA boundaries was considered highly impacted (USEPA 2003a).

Table 2-7. Historical Benthic Community Indices from REMAP

Metric	REMAP Study Year					BERA
	1993/94	1998	2003	2008	2013	2015
Number of NBSA Stations	**18**	**13**	**11**	**11**	**11**	**30**
Species Richness						
Number of Species	73	53	73	71	22	106
Means of Benthic Variables (± 90% confidence interval)						
Species Richness (as number species/sample)	19 (±2.5)	14 (±2.4)	22 (±4.8)	22 (±3.5)	6.7 (±1.2)	23 (±2.9)
Pollution-Sensitive Species (%)	0.5 (±0.5)	1.7 (±1.8)	2.4 (±2.2)	3.5 (±3.4)	0	1.7 (±0.8)
Pollution-Indicative Species (%)	57 (±7.5)	47 (±13)	42 (±8.5)	20 (±9.2)	41 (±12)	31.2 (±4.9)

Species Diversity (Shannon-Wiener)*	1.5 (±0.14)	1.8 (±0.21)	1.8 (±0.19)	1.9 (±0.13)	1.3 (±0.16)	2.2 (±0.10)

Notes:
* Natural Log Calculation
All values calculated from raw data.
NBSA = Newark Bay Study Area
SQT = sediment quality triad

More recent studies on infaunal benthic communities conducted during the period of the NBSA RI—which include the 2008 and 2013 REMAP studies, and surveys conducted in 2015 as part of the SQT and Porewater Study (Tierra 2017b) for this BERA—have shown a substantial improvement in benthic community conditions in the Bay (Figure 2-4). An apparent downward trend in the species diversity (richness) and B-IBI values for the 2013 REMAP dataset may be due to the short-term effects of Superstorm Sandy on benthic communities. The values from the 2015 program show a return to the relatively unimpacted conditions similar to those from the 2008 REMAP study.

In the 2015 SQT and porewater investigation, a cumulative total of 106 individual taxa were found throughout the Bay, with a range between 8 and 46 taxa identified per individual sample location. The ten most abundant taxa are compared to historical abundant taxa on Table 2-6. A comparison of benthic community indices from the BERA (SQT) dataset to those from the REMAP program are provided in Table 2-7. In each program, specimens were identified to the lowest practicable taxonomic level, usually genus and species (Tierra 2015b; USEPA 2003a).

Similar to previous investigations, polychaetes, amphipods, and small clams continue to dominate the species composition. Metrics on the diversity and abundance of invertebrates have also improved (i.e., more diversity and evenness of species found) in recent years (Table 2-8). Shannon-Wiener Diversity values from the 2015 surveys ranged from 1.5 and 3.0, with an overall mean of 2.2 (n = 30). And community density calculations produced values ranging between 406 and 11,237 individuals per square meter, with a mean of 4,771 (n = 30). Compared to historical REMAP studies, the B-IBI for the 2015 surveys indicates as much as 70% of the Newark Bay benthos are considered to be not impacted, with a single station considered highly impacted (Table 2-9; Figure 2-4). Improvements to benthic community conditions are likely due to improving conditions in the Bay in terms of water and sediment quality. Improvements in the incidence of acute toxicity based on 10-day amphipod survival have also been observed. In a toxicity study conducted by NOAA (1995), 88% of samples in the NBSA were toxic. The percentage of toxic samples generally declined over the years of the REMAP studies: 22% in 1993/1994, 54% in 1998, 9% in 2003, 9% in 2008, and 0% in 2013 (see Appendix A). A more detailed evaluation of the benthic invertebrate community metrics relative to sediment chemistry and toxicity is provided in the SQT assessment (Appendix A) and Section 6.

2.3.2 Shellfish

Historically, communities of large invertebrates in the channels and shoals of the Bay are mainly dominated by blue crab (*Callinectes sapidus*), which are present year-round in the Bay. Blue crabs at shallow water stations (subtidal flats) appear to be more abundant in the summer and fall (May to November) and are nearly absent during the other months when they migrate to the deeper channels or offshore (USACE 1997). Bivalve mollusks in the Bay include the softshell clam (*Mya arenaria*), dwarf

surfclam (*Mulinia lateralis*), ribbed mussel (*Geukensia demissa*), Eastern oyster (*Crassostrea virginica*), and Baltic clam (USACE 1997).

As part of the 2013 reconnaissance survey, a sampling program was conducted to evaluate the locations in the NBSA where bivalves were present to establish formal target sampling locations for the subsequent 2014 clam and crab sampling program. The bivalve collection results from various intertidal survey locations confirmed the presence of softshell clam, ribbed mussel, Eastern oyster, and Baltic clam in Bay sediments. Softshell clams that were found, which represents the target species for this BERA sampling, were counted and measured, then returned to the sediments.

Blue crab and softshell clam collection surveys in support of this BERA were collected in late summer/early fall 2014 throughout the NBSA. Blue crabs were collected from 12 intertidal locations and an additional 25 locations throughout the three geographic zones of the Bay and softshell clams were collected from 18 intertidal sample locations (Figure 2-5). Additional details of the sampling program can be found in the NBSA Crab and Clam Data Report (GSH 2017b) and the NBSA *Environmental Sample Collection Report* (Normandeau 2017). The latter is provided as Attachment 4 to this BERA.

A total of 312 blue crab were collected during the course of the intertidal area collection effort with 68% of individuals being male. An additional 481 blue crab were collected during the geographic collection efforts, with 79% of the individuals identified as male. Carapace width among all individuals collected ranged between 112 and 198 millimeters (mm; 4.4 and 7.8 inches) with an overall mean of 139 mm (5.5 inches) (Normandeau 2017). A total of 1,159 softshell clam were collected with a mean shell length of 49.3 mm (1.9 inches) and range between 31.0 and 92.0 mm (1.2 and 3.6 inches) (Normandeau 2017). Softshell clam lengths and blue crab carapace lengths from other comparable estuaries are summarized in Tables 2-10 and 2-11. The size of the softshell clams collected in the NBSA may be indicative of young (< ~~2years~~2 year old) clams when compared to the sizes in mid-Atlantic estuaries (Table 2-10). The range of shell lengths of the sampled softshell clam population could be due to various environmental factors, including (but not limited to) the specific habitat characteristics of the Bay intertidal and shallow subtidal areas where softshell clam samples were collected, competition with other clam species, predation by upper trophic level organisms, pathogens, and/or exposure to chemical contaminants. The size of blue crabs collected in the NBSA appear to indicate that they are comparable in size to adult crabs from other mid-Atlantic estuaries (Table 2-11).

Table 2-10. Size Ranges of Softshell Clam from Other Estuaries

Location	Life Stage	Size Class	Softshell Clam Length (millimeters)	Source
Mid-Atlantic	Adult (5+ years)	Average	75-100	Abraham and Dillon 1986
Mid-Atlantic	Adult (~ 8 years)	Max	150	Abraham and Dillon 1986
Mid-Atlantic	Juvenile (late veliger stage)	Average	0.2	Abraham and Dillon 1986
Mid-Atlantic	Juvenile (before burrowing)	Max	12	Abraham and Dillon 1986
Mid-Atlantic	Juvenile (first winter)	Average	30	Abraham and Dillon 1986
Mid-Atlantic	Young (1.5 - 2 years)	Minimum commercial	50	Abraham and Dillon 1986

Table 2-11. Size Ranges of Blue Crab from Other Estuaries

Location	Life Stage	Size Class	Carapace Width (millimeters)	Source
Mid-Atlantic	Second larval stage (megalops)	Average	2.5	Hill et al. 1989
Mid-Atlantic	Mature females	Min	55	Hill et al. 1989
Mid-Atlantic	Mature females	Max	204	Hill et al. 1989
Mid-Atlantic	Mature males	Max	209	Hill et al. 1989
Chesapeake Bay	Adult	Average	178	Hill et al. 1989; Van Den Avyle 1984
Chesapeake Bay	4 months	Average	64	Hill et al. 1989; Van Den Avyle 1984
Chesapeake Bay	12-16 months	Average	127	Hill et al. 1989; Van Den Avyle 1984
South-Atlantic	Mature females	Min	51	Van Den Avyle 1984
North and South Carolina	Adult	Average	203	Van Den Avyle 1984
Florida	Immature females	Max	177	Van Den Avyle 1984
St. Johns River, FL	Adult	Min	102	Van Den Avyle 1984
St. Johns River, FL	Adult	Max	178	Van Den Avyle 1984

2.4 Fish

The finfish assemblage that resides in or is transient to the Bay is typical of large coastal estuaries and inshore waterways located along the Mid-Atlantic Bight. Situated in the transition zone between northern coldwater (boreal) and temperate (warm) water with low to moderate salinity, the Bay acts as a spawning ground, migratory pathway, and a nursery/foraging area for a variety of estuarine, marine, and anadromous fish species.

An essential fish habitat (EFH) designation exists in the Bay for several finfish species (USACE 2013a). The entire Bay is identified as EFH for one or more important life stages (i.e., eggs, larvae, juveniles, adults, and spawning adults) of 19 fish species (Table 2-5). The shortnose and Atlantic sturgeon are federally listed as protected species in the NBSA (Tierra 2015a; USACE 2013a). The latest EFH data, as presented in Table 2-5, was extracted from NOAA's online EFH data mapper for the Hudson River/Raritan/Sandy Hook Bays area (http://www.nero.noaa.gov/hcd/est.htm).

Several species found in the NBSA are designated NOAA Species of Concern (SOC), including alewife (*Alosa pseudoharengus*), blueback herring (*Alosa aestivalis*), and rainbow smelt (*Osmerus mordax*). SOCs are species for which NOAA's National Marine Fisheries Service has concerns regarding danger of extinction or risk of becoming endangered but for which insufficient information is available to indicate a need to list. Alewife and blueback herring are obligate estuarine-dependent species commonly found in NY Harbor, while rainbow smelt are diadramous fish that are relatively uncommon in NY Harbor (USACE 2015a, 2015b).

Many of the seasonally abundant fish species in the Bay are transient or migratory, passing through the Bay to upstream spawning grounds or entering the area seasonally from nearby ocean waters. These include estuarine migratory species, such as striped bass (*Morone saxatillis*), that depend on the estuary as a nursery and a forage area for juveniles and adults. Species that frequent the Bay during similar life history stages include both marine and estuarine predators, such as winter flounder (*Pseudopluronectes americanus*), bluefish (*Pomatomus saltatrix*), and summer flounder (*Paralichthys dentatus*). These fish migrate in and out of the Bay seasonally depending on spawning area (i.e., marine, estuarine, or freshwater) and season (USFWS 1997; Woodhead 1991).

A few fish species, primarily forage fish such as the mummichog (*Fundulus heteroclitus*) and striped killifish (*Fundulus majalis*), are year-round residents in the Bay. These species generally begin spawning in late spring and continue throughout most of the summer. Other forage fish species in Newark Bay include bay anchovy (*Anchoa mitchilli*), Atlantic silverside (*Menidia menidia*), and the river herring species alewife and blueback herring. Bay anchovy, alewife, and blueback herring are very common in Newark Bay; Atlantic silverside are present in smaller numbers (USACE 2015b). Bay anchovy, a facultative estuarine-dependent species, is likely resident in the Bay and is most common in spring. Blueback herring and alewife are anadramous, entering estuaries in the spring and migrating to freshwater to spawn. Atlantic silverside are found in estuarine and near-shore habitats and are most common in the Bay in winter (USACE 2105b). These species provide an important forage base for larger predatory species.

2.4.1 Newark Bay Fishery Characterization

Several fisheries surveys have been conducted in Newark Bay since the 1990s. The goals and sampling approach of these surveys differed but, together, they provide a sound information base to characterize the seasonal distribution and composition of the fish community in various habitats and areas of the Bay.

The NMFS conducted a major fish sampling program in Newark Bay as part of an evaluation of a flood control project for the Passaic River Basin (NOAA 1994a). This study provided information on the habitat preferences of species (e.g., channel vs. shoal [subtidal] areas). Monthly fish sampling was conducted from May 1993 through April 1994. Juvenile and adult fish were sampled with bottom trawls and gill nets. A total of 56 species of fish and invertebrates were identified from 299 otter trawl tows, 105 shrimp trawl tows, and 92 gill net sets. The dominant species caught in the channels were striped bass and Atlantic tomcod (*Microgadus tomcod*). The dominant species caught in the shrimp trawl were bay anchovy (*Anchoa mitchilli*), Atlantic herring (*Clupea harengus*), and Atlantic tomcod. The dominant species caught in the gill nets were Atlantic menhaden (*Brevooria tyrannus*) and striped bass. From the seven shoal stations on the east side of Newark Bay (relative to the main channel), 28 species of fish were collected with a bottom trawl. Six species – striped bass, winter flounder, bay anchovy, Atlantic herring, Atlantic

tomcod, and Atlantic silverside (*Menidia menidia*) dominated the catch from all shoal stations combined (NOAA 1994a). Most species, whether resident or transient, were found throughout Newark Bay. The notable exception is white perch (*Morone americana*~~), which was not collected in waters that had less than 6.23 milligrams per liter (mg/L) of dissolved oxygen (DO) (NOAA 1994b). Across all sampling stations (including shallow water and channel stations) in Newark Bay during this study, DO levels varied from an average of 3.79 mg/L in August to 10.57 mg/L in March.~~). Few to no white perch were collected in Newark Bay during July and August, likely because they migrate into streams from Newark Bay in spring to begin spawning and remain there until fall when they return to winter in the deeper waters of Newark Bay (Stanley and Danie 1983).~~ The DO levels did not appear to affect the presence of the other most abundance species in Newark Bay (NOAA 1994b).~~

In order to describe the species composition and relative abundance of fish in shoal areas of Newark Bay, Lawler, Matusky and Skelly Engineers, Inc. conducted sampling trawls from April 1995 to March 1996 (Lawler, Matusky and Skelly Engineers, Inc. 1996). Four shoal areas in Newark Bay were sampled to provide information on the fish community. Species collected from the shoal stations were dominated by relatively few species. Eight species – bay anchovy, striped bass, winter flounder, windowpane flounder (*Scophthalmus aquosus*), Atlantic silverside, summer flounder, northern pipefish (*Syngnathus fuscus*), and white perch – dominated the catch. Most species occurred infrequently and in very low numbers during the 12-month study. However, there was a consistent seasonal pattern for fish among the shoal stations. Fish were relatively abundant from April through October, but much less abundant from November through March. During the period when fish were most abundant, only four species—striped bass, winter flounder, summer flounder, and bay anchovy—occurred during each month.

The USACE collected fish community data from within the Bay annually between 2001 and 2011, using a 30-foot otter trawl, as part of the NY/NJ Harbor Deepening Project (HDP). Both bottom and mid-water trawl sampling was conducted to evaluate demersal and pelagic species assemblages. Together with the earlier datasets described above, these surveys provide a valuable long-term dataset to assess the consistency of fish communities amid changing conditions and anthropogenic alterations in the Harbor. The USACE data are reported in annual Aquatic Biological Survey Reports from 2001 through 2011 (USACE 2002, 2003, 2004c, 2005, 2006b, 2007c, 2008, 2009b, 2010, 2011, 2013b) and summarized in the USACE's *Migratory Finfish Summary Report* (2015). The latter report primarily focuses on the results of mid-water trawls during migratory finfish sampling in 2006, 2011, 2012, and 2013. The number of trawls conducted during each of these years throughout the Bay ranged from 311 to 440, with 23 to 38 finfish species collected in any given year. In Newark Bay, species abundance has varied from year to year, but is generally dominated by a few species (e.g., white perch, striped bass, and bay anchovy).

The aggregate available fish community data collected by the USACE in bottom trawls from Newark Bay between 2002 and 2010 are summarized in Figures 2-6a (overall species composition of multi-year catch) and 2-6b (average monthly composition of multi-year catch). The seasonal data from mid-water trawls collected between 2006 and 2013 are summarized in Figure 2-7. Numbers of some species, such as American eel (*Anquilla rostrata*) and mummichog, may be under-represented in these studies due to the types of fishing methods used. Success rates for capturing such species may be higher with appropriate minnow or eel traps, rather than tow/otter trawls or gill nets.

As shown on Figure 2-6a, during nine years of sampling, the following nine species dominated the overall composition of Newark Bay finfish community: white perch (44%); bay anchovy (13%); Atlantic herring

(13%); striped bass (12%); spotted hake (8%); winter flounder (3%); red hake (*Urophycis chuss*) (2%); alewife (*Alosa pseudoharengus*) (2%); blueback herring (*Alosa aestivalis*) (2%). The remaining species each represented less than 1% of the catch. The shortnose sturgeon (*Acipenser brevirostrum*), although federally listed as a protected species in Newark Bay, has not been documented in any of the studies in Newark Bay or adjacent waters (USACE 1997).

The seasonality of the finfish community is shown in Figures 2-6b (bottom trawl sampling) and 2-7 (mid-water trawl sampling). Bottom trawl sampling shows that white perch and striped bass populations peak in the winter and decline in the summer. Both Atlantic herring and spotted hake peak in spring and bay anchovy peaks in the summer. Mid-water trawl data also show the large increase in bay anchovy composition in summer and fall.

A major ichthyoplankton survey conducted in Newark Bay between 1993 and 1994 by the National Marine Fisheries Service collected larvae of 20 different species of fish. Only two species, bay anchovy and one unidentified goby (*Gobiosoma* sp.), were collected in substantial numbers at any time during the study year. Both species were present from June through September. More thorough ichthyoplankton surveys were also conducted by USACE from seven stations within Newark Bay during sampling events for the HDP (USACE 2011). Data collected between 1999 and 2006 (USACE 2003, 2004c, 2005, 2006b) are presented in Table 2-12. Forty-two species of ichthyoplankton were identified at various life stages (i.e., egg, larvae, or juvenile), including eggs and larvae from unidentified species These surveys targeted the egg and larval life stages and were conducted on a seasonal basis, primarily from January to June each year, therefore some species, as well as the juvenile life stage, may be under-represented in this dataset. Bay anchovy dominated the catch across all 8 years. All life stages were captured for bay anchovy, winter flounder, weakfish, and windowpane flounder, indicating that these species are spawning and have early life stages in Newark Bay.

2.4.2 NBSA Fish Community Characterization: 2014 - 2016

A fish sampling program was implemented as part of the NBSA BERA and BHHRA sampling activities. The objective of this program was to collect fish tissue samples from various target species at different trophic levels throughout the Bay to support the NBSA BERA and BHHRA.

A quantitative fish community characterization was not conducted under this program, but both target and non-target species captured as part of these sampling efforts were identified, counted, measured (length) and weighed. Given the substantial fishing efforts associated with this program, the various gear types used, and the different habitat sampled (i.e., intertidal, shallow subtidal, deepwater channels), these data provide additional perspectives to the historical studies on the species assemblages inhabiting or utilizing Newark Bay during different seasons. The fish collected were also evaluated for their apparent health and condition by a fish pathologist, as discussed in Section 7.

Under the BERA/BHHRA sampling program, fish were collected and identified over three sampling events in fall 2014, spring/summer 2015, and spring 2016. During these fish sampling events, 43 fish species were collected. A listing of the identified species is provided in Table 2-13. Additionally, three species of ~~crab~~decapod crustaceans (blue, mud, and spider crabs), as well as horseshoe crab (a ~~decapod crustacean~~chelicerate arthropod) were collected as bycatch; blue crab was the most abundant species collected. Additional details of the sampling program can be found in the NBSA Fish Data Report (Tierra 2017) and Normandeau (2017) (Attachment 4).

Fish sampling activities were conducted within the three Newark Bay geographic zones (north, central, and south) during three fish sampling events: fall 2014, spring/summer 2015, and spring 2016. Fish were collected from various locations along the subtidal flats, transitional slopes, and navigational channel throughout each of the three zones in the NBSA. Fish were collected from each of the different geographic areas to provide spatial coverage across the entire NBSA. Figures 2-8, 2-9, and 2-10 present the various fish sampling locations throughout the NBSA.

Additionally, 12 intertidal areas were targeted for up to 19 forage fish samples (mummichog, striped killifish, Atlantic menhaden, or bay anchovy). These areas are consistent with those selected for clam and sediment sampling (refer to the Crab and Clam QAPP; Tierra 2014a). Forage fish were targeted at 16 sampling locations and three supplemental sampling locations (Intertidal Areas 124, 125, and 126). Additional details regarding forage fish collected during this event are provided in the Normandeau (2017).

Fish sampling methods employed during each of the three sampling events include at least one of the following: trawls, gillnets, eel pots, minnow traps, seine, traditional angling, and trotlines. A summary of sampling method by event is provided in Table 2-14. Additional details regarding sampling methods used during fish sampling activities are provided in the Normandeau (2017).

Table 2-14. Fish Sampling Method Summary

Sampling Method	Fall 2014	Spring/Summer 2015	Spring 2016
Trawl	X	X	X
Gillnet	X	X	X
Eel pot	X	X	
Minnow Trap	X		
Seine	X		
Angling		X	
Trotline		X	

The fall 2014 fish sampling event was conducted from October 6 through 24, 2014. Thirty-one fish species, represented by 6,454 individuals, were collected during this event. Sampling locations are presented on Figure 2-8. Tables 2-15 and 2-16 summarize target and non-target fish species collected during the fall 2014 fish sampling event, respectively. Additional details regarding the fish collected during the fall 2014 fish sampling event are provided in Normandeau (2017).

The spring/summer 2015 fish sampling event was conducted from March 30 through August 5, 2015. Four separate collection efforts were conducted March 30 through April 10, April 27 through May 8, June 1 through June 8, and July 27 through August 5, 2015, each targeting specific fish species. Thirty fish species, represented by 2,243 individuals, were collected during this event. Sampling locations are presented on Figure 2-9. Tables 2-17 and 2-18 summarize target and non-target fish species collected

during the spring/summer 2015 fish sampling event, respectively. Additional details regarding the fish collected during the spring/summer 2015 fish sampling event are provided in Normandeau (2017).

Due to an error during homogenization of white perch tissue samples at the analytical laboratory in late 2015, a fish sampling event was conducted from March 14 through 24, 2016, targeting collection of white perch to compensate for the compromised samples. Twenty-five fish species, represented by 3,821 individuals, were collected during the spring 2016 fish sampling event. Sampling locations are presented on Figure 2-10. Tables 2-19 and 2-20 summarize target and non-target fish species collected during the spring 2016 fish sampling event, respectively. Additional details regarding the fish collected during the spring 2016 fish sampling event are provided in Normandeau (2017).

Overall, the 2014-16 BERA fish sampling program confirmed the results of the more formal fisheries surveys—i.e., that the Newark Bay fishery is primarily migratory, with a variety of species occurring on a seasonal basis. Of particular note, on March 17, 2016 the fish sampling yielded an incidental capture of a single Atlantic Sturgeon. This species of fish is listed as endangered. The fish was captured in a gillnet set on the bottom of Newark Bay in approximately 40 feet of water. The fish appeared to be in good condition and was identified and returned, unharmed, to the water. Additional information regarding the incidental capture of this fish is provided in the NBSA Environmental Sample Collection Report (Normandeau 2017).

2.5 Birds

Despite the highly developed shoreline, the NBSA provides various habitats for birds. Populations of birds that inhabit or utilize the forested (and other) areas of the Bay are mainly water birds, including piscivorous species like cormorants (Phalacrocoracidae), osprey (*Pandion haliaetus*), egrets, herons, and terns; omnivorous species including ducks, geese; sediment-probing invertivorous species including sandpipers (Scolopacidae), plovers (*Charadrius melodus*); and other invertivores such as oystercatchers (Haematopus).

Sediment-probing invertivores and piscivorous wading birds like egrets and herons can be found foraging in mudflats along the Lower Passaic and Hackensack Rivers and in tidal marshes within the region, particularly the Hackensack Meadowlands and marshes to the south of Newark Bay adjacent to the Arthur Kill and Staten Island (Iannuzzi and Ludwig 2004; Ludwig et al. 2010; Windward Environmental LLC [Windward] 2011). They are not expected to be a major component of the bird fauna of the relatively restricted confines of Newark Bay, but they do likely forage on the few intertidal mudflats of Newark Bay. Piscivorous species such as cormorant forage in the open waters of the NBSA. Omnivorous bird species, primarily waterfowl, use the subtidal areas and open waters of the NBSA for foraging and loafing.

Many of the water bird species found in the Newark Bay region (and NY/NJ Harbor Estuary as a whole) are migratory. The area is part of the Atlantic Flyway, a major north-south migration route that is used by migrant species as a seasonal stopover (Elphick et al. 2001). The birds utilize open waters of the Bay and Intertidal Areas, feeding and resting for a few days to a few weeks in route to northern breeding grounds or southern wintering areas. Sandpipers, plovers, and their relatives are abundant migrants, and some are winter residents (Walsh et al. 1999). Bird species observed in the NBSA from several bird surveys that have been conducted in the region are identified in Table 2-21.

Some species, including mallard (*Anas platyrhynchos*), American black duck (*Anas rubripes*), gadwall (*Anas strepera*), and double-crested cormorant (Walsh et al. 1999) are known to breed in the NBSA: however, adequate nesting grounds are limited due to the highly urbanized shoreline of the Bay and its tributaries (i.e., the Passaic River, Hackensack River, Arthur Kill, and Kill van Kull). Therefore, most water birds primarily build their nests for breeding offshore on the 17 islands located within the NY/NJ Harbor Estuary. Two of the islands are located in the Arthur Kill/Kill van Kull complex (Prall's Island and Isle of Meadows), while Shooters Island is located in the southern portion of the Bay. These islands and, to a lesser extent, mainland areas of the NY/NJ Harbor Estuary, have been surveyed for decades to quantify and characterize the nesting and trends in water bird populations.

During the 2013 Reconnaissance Survey, no bird nests were observed along any portions of the shoreline; rather, they were observed/located on structures in the water (e.g., mile markers, abandoned/dilapidated wooden and metal pier structures). Both large and small nests were observed during the survey, which was limited to shoreline areas where field biologists were able to complete survey activities on foot, while looking for bird nests and signs of mammal use (e.g., scat, tracks). The large nests consisting of hard sticks are likely to be osprey nests. The smaller nests consisting of softer grasses and seaweed that are also located on structures in the NBSA are likely to be double-crested cormorant (*Phalacrocorax auritus*) nests. Bird nest locations observed during the limited survey are shown on Figure 2-11 (Tierra 2015a).

2.6 Mammals

Several primarily omnivorous and herbivorous mammals (primarily urban opportunistic species, such as rats and coyotes) were observed throughout the NBSA during the Reconnaissance Survey. Direct mammal sightings were documented, as well as any signs of mammals present along the shoreline (e.g., tracks, scat). Most mammals observed are species that are frequently found in urban areas, including Norway rat (*Rattus norvegicus*), skunk (e.g., *Mephitis mephitis*), coyote (*Canis latrans*), deer (e.g., *Odocoileus virginianus*), rabbit (e.g., *Sylvilagus* spp.), cat (*Felis catus*), and muskrat (*Ondatra zibethicus*) (Tierra 2015a). Locations of mammal observations observed during the limited survey are shown on Figure 2-11.

USACE (1997) identified nine species of bats, a muskrat ~~(*Ondatra zibethicus*),~~, and a harbor seal as "possible" mammalian species utilizing the Subtidal Flats of Newark Bay. There have been recent sightings of harbor seals (*Phoca vitulina*) and harp seals (*Pagophilus groenlandicus*) in the Hackensack River (New Jersey Meadowlands Commission 2010, 2011; Frazier 2011; Baldwin 2018). These seals would have had to traverse Newark Bay to get to foraging grounds in the Meadowlands. Although they are not common in the NBSA, they are a potential visitor.

Several primarily piscivorous mammal species may potentially use the NBSA. The river otter is a potential inhabitant of the marine reaches of NY/NJ Harbor Estuary. Although commonly called a "river" otter, the name can be misleading, as the animal inhabits marine, as well as freshwater environments, and some populations permanently reside in marine shoreline habitats. The North American river otter is found in a wide variety of aquatic habitats, both freshwater and coastal marine, including lakes, rivers, inland wetlands, coastal shorelines and marshes, and estuaries. The river otter's main requirements are a steady food supply and easy access to a body of water. The North American river otter has been

extirpated from much of its historical range due to a number of factors including habitat loss and pollution (Dewey and Ellis 2003).

Piscivorous harbor seals and gray seals are observed in the New York Bight. They winter in New York Harbor and haul out onto islands in Staten Island, Long Island, Governor's Island, Manhattan, Sandy Hook, Jamaica Bay, the Westchester and Connecticut shorelines of Long Island Sound, as well as more recently on Hoffman and Swinburne Islands, two small artificial islands off the coast of Staten Island (NJ Sea Grant 2014). Harbor seal have been sighted in the Arthur Kill as well as the Hackensack River in recent years but are generally uncommon visitors to Newark Bay (USACE 1997) and observed rarely in the Hackensack Meadowlands (Kiviat and MacDonald 2002).

Other marine mammals, such as cetaceans, may potentially be found near Newark Bay. Whales are rarely observed in Newark Bay; one baleen whale species was found dead in Newark Bay in 1975 (NY Times 1975); one humpback whale (*Megaptera novaeangliae*) was observed in the Hackensack River and Newark Bay in 1990 (Murphy 1990), and one fin whale (*Balaenoptera physalus*) was struck and killed by a boat in Newark Bay in in 2016 (Beckford 2016). Whales have also been observed in the New York Bight and New York Harbor (New York State Department of Environmental Conservation [NYSDEC] 2019). Bottlenose dolphins (*Tursiops* ~~*truncates*~~*truncatus*) have also been observed in New York New Jersey Harbor and the Hackensack River (New York State Department of Environmental Conservation [NYSDEC] 2019).

There is limited available aquatic habitat for these marine mammals in the NBSA. In addition, the noise and traffic of cargo ships entering and leaving Newark Bay would likely deter these animals from intentionally entering it. As discussed by Battelle (2008), the possibility of cetaceans entering Newark Bay is remote.

2.7 Amphibians and Reptiles

In freshwater wetlands, reptiles and amphibians are typically abundant. However, given the estuarine nature of the water of Newark Bay, amphibians, such as salamanders and frogs, are unlikely to be present (USFWS 1997). During the 2013 Reconnaissance Survey, the only reptile observed was a diamondback terrapin (*Malaclemys terrapin*) along an area of emergent vegetation (Tierra 2015a). The diamondback terrapin is likely a transient visitor to the NBSA as it moves between the Hackensack Meadowlands Complex and marshes and tidal creeks along the Arthur Kill (USACE 1997). Other than the observation of the terrapin noted above during the Reconnaissance Survey, there are no documented observations of amphibians or reptiles using the Bay during the various ecological surveys that have been performed over the years.

USACE (1997) also notes the possibility for three species of marine turtles – loggerhead (*Caretta caretta*), Kemp's ridley (*Lepidochelys kempii*), and the green sea turtle (*Chelonias mydas*) – to utilize the shoal areas (i.e., subtidal flats) of Newark Bay. However, there is no documented evidence of these species in the NBSA.

2.8 Threatened, Endangered, and Special Status Species

The state listed threatened and endangered species consist mainly of birds, but also include three insects, one plant, and one fish species. The State of NY also lists one habitat. The only federally endangered species are the Atlantic and shortnose sturgeon. (*Acipenser oxyrhynchus* and *brevirostrum*, respectively). The species and habitat are identified in Table 2-22.

To identify these species, a request was submitted to the National Heritage Program for the State of New Jersey, and an online search using the NYSDEC Nature Explorer was conducted with a user-defined subject area for the State of New York. The online search area consisted of the southern portion of Newark Bay and portions of the Kills, which are the areas of Newark Bay that fall under New York state jurisdiction.

Although the diamondback terrapin (*Malaclemys terrapin*) is not currently listed on federal or state-level threatened and endangered species lists, it has been under review for possibly listing (NJDEP 2016) and is found in the Hackensack River watershed.

Table 2-~~20~~22. Threatened and Endangered Species in the NBSA

Species	Common Name	Federally Listed	State Listed
Birds			
Nycticorax	Black-crowned night heron		NJ
Bubulcus ibis	Cattle egret		NJ, NY
Plegadis falcinellus	Glossy ibis		NJ, NY
Sterna antillarum	Least tern		NJ
Egretta caerulea	Little blue heron		NJ, NY
Pandion haliaetus	Osprey		NJ
Falco peregrinus	Peregrine falcon		NJ
Egretta thula	Snowy egret		NJ, NY
Egretta tricolor	Tricolored heron		NJ
Nyctanassa violacea	Yellow-crowned night heron		NJ, NY
Ardea alba	Great egret		NY
Tyto alba	Barn owl		NY
Insects			
Pontia protodice	Checkered white butterfly		NJ
Somatochlora linearis	Mocha emerald dragonfly		NY
Ischnura ramburii	Rambur's forktail damselfly		NY
Plants			
Elocharis quadrangulata	Angled spikerush		NY

Species	Common Name	Federally Listed	State Listed
Fish			
Acipenser brevirostrum	Shortnose sturgeon	X	NJ
Acipenser oxyrhynchus	Atlantic sturgeon	X	
Habitats			
Colonial Waterbird Nesting Area			**NY**

3 PROBLEM FORMULATION

The USEPA Superfund eight-step process for conducting ERAs is depicted on Figure 3-1 (USEPA 1997). The Problem Formulation (Step 3) for the NBSA BERA was completed in 2013 and is provided as Attachment 2 to this BERA (Tierra 2013a). As part of the Problem Formulation process, the following activities were conducted:

- Refined the preliminary list of constituents of potential ecological concern (COPECs)
- Preliminarily characterized the potential ecological effects of COPECs
- Reviewed and refined information on the fate and transport of COPECs, potential exposure pathways, and receptors potentially at risk
- Selected assessment and measurement endpoints
- Refined the ecological conceptual site model (CSM) with testable hypotheses (i.e., risk questions) that this BERA will address.

This BERA consists of Steps 3 through 7 of the USEPA's eight-step ERA process (Figure 3-1; USEPA 1997) and builds on the results of the SLERA (Battelle 2008) to provide a more accurate and realistic estimate of potential ecological risk in the NBSA. To do this, the Problem Formulation was used to guide the collection of site-specific data for use in estimating potential risks to invertebrates, fish, and wildlife. A risk analysis plan is also included in this section that presents the various lines of evidence (LOEs) and overall weight of evidence (WOE) process that was used to draw risk conclusions in the BERA. Results of the BERA will then be used to inform the remedial action decision-making process (Step 8 of the risk assessment process).

3.1 Assessment and Measurement Endpoints

Assessment endpoints (AEs) are defined as "an explicit expression of the environmental value that is to be protected" (USEPA 1997). Because it is not practical or possible to directly evaluate risks to all of the individual components of the ecosystem, AEs focus the risk assessment on particular components of the ecosystem that could be adversely affected by site-related constituents. The AEs were selected based on various discussions and document exchanges among Tierra, the USEPA, and the Partner Agencies, as summarized in the Problem Formulation (Tierra 2013a) and refined in the November 2015 Technical Memorandum entitled *Risk Assessment Field Sampling and Analysis Program – Newark Bay Study Area* (Arcadis 2015). They include:

1. Survival, growth, and/or reproduction of invertebrates
2. Survival, growth, and/or reproduction of fish
3. Survival, growth, and/or reproduction of birds
4. Survival, growth, and/or reproduction of mammals
5. Survival, growth, and/or reproduction of reptiles.

Following the selection of AEs, testable hypotheses are developed to determine whether or not a potential risk to the AE exists (USEPA 1997). A testable hypothesis is an operational statement of an investigator's research assumption made to evaluate logical or empirical consequences (USEPA 1997, 1998). Similar to the LPRSA, the testable hypotheses for the NBSA are presented as a series of risk questions about the relationship between each of the AEs and the responses of the receptors when exposed to chemicals within Newark Bay. Hypotheses usually postulate that there is no effect or no difference (among groups or measurements), and data are collected to confirm or refute that hypothesis. Consistent with USEPA guidance on the formulation of hypothesis tests (USEPA 2006a, 2006b), all hypothesis tests comparing site metrics to reference metrics were conducted as one-sided tests where the alternative hypothesis is that the site metric demonstrates degradation. A potential consequence of hypothesis testing is a Type II error, (i.e., accepting the null hypothesis, when it is, in fact, false). Adequate sample size is the best way to avoid such an error.

A measurement endpoint (ME) is defined as the "measurable ecological characteristic that is related to the valued characteristic chosen as the assessment endpoint" (USEPA 1997). Each ME represents a LOE that, along with consideration of the LOE strengths and uncertainties, is used in the WOE approach to draw risk conclusions. In some cases, additional LOE were available (e.g., site-specific toxicity data from the published literature), which were also included in the WOE. Final MEs were selected based on various discussions and document exchanges among Tierra, the USEPA, and the Partner Agencies, as summarized in the Problem Formulation (Tierra 2013a) and refined in Arcadis (2015).

Table 3-1 presents the final selected AEs, MEs, risk questions, and a brief summary of data collected to support the BERA. In addition to the identified AEs, the BERA addresses the perspectives on potential risk from sediment or water exposures to other organisms, including plankton, algae, and aquatic vegetation. These are discussed in Section 11. The following subsections describe the AEs and MEs by receptor group in more detail.

3.1.1 Invertebrates

Assessment Endpoint 1: Survival, growth, and/or reproduction of invertebrates.

> Invertebrates form the base of the food web for fish and other upper-trophic level organisms. There are a variety of taxonomic feeding guilds for aquatic and benthic invertebrates including filter-feeders, deposit-feeders, detritivores, and omnivores. Each taxonomic feeding guild has a unique exposure potential to COPECs in NBSA media that is evaluated in the BERA.

Risk Question: *Are invertebrate communities in the NBSA different from those found in similar nearby water bodies with chemical concentrations at regional background levels?*

> This question is addressed by comparing community structure data (e.g., total invertebrate abundance, species richness, and abundance of species or specific taxonomic groups) from Newark Bay to reference populations from Jamaica Bay via statistical comparisons of community indices. This LOE is part of the SQT assessment (Appendix A; Section 6), which is a sediment assessment technique that incorporates information about sediment chemistry, toxicity, and benthic community metrics.

Risk Question: *Are COPEC residues in invertebrate tissues from the NBSA greater than tissue (whole-body) toxicity benchmarks for the survival, growth, and/or reproduction of invertebrates?*

This question is addressed by comparing chemical concentrations in laboratory-exposed and/or site-collected invertebrate tissues to literature-based toxicity reference values (TRVs). The data use objective for this ME is to assess the adverse effects of COPECs on the invertebrate community. Field-collected tissue data from softshell clams and blue crabs from the NBSA are analyzed and compared to literature-based TRVs. In addition, data from the limited number of polychaete worm (*Nereis virens*) bioaccumulation tests conducted are also compared to available TRVs.

Risk Question: *Are the COPEC concentrations in sediments from the BAZ greater than benchmarks for the survival, growth, and/or reproduction of invertebrates?*

Chemical concentrations in sediment are compared to toxicological sediment benchmarks from the literature. The data use objective for this ME is to evaluate the effects of chemical concentrations in sediment on the benthic invertebrate community of the NBSA. Surficial sediment was collected from the BAZ, which is estimated to be the top 6 inches, and chemically analyzed. This LOE is part of the SQT assessment (Appendix A; Section 6).

Risk Question: *Is the survival, growth, and/or reproduction of invertebrates exposed to sediments from the BAZ of the NBSA significantly lower than that in reference sediments?*

Laboratory toxicity tests using NBSA surface sediment were conducted, including 10-day and 28-day studies with *Leptocheirus plumulosus* for survival, growth, and reproduction. Amphipods are considered to be a sensitive biological organism for representing potential risk to the benthic community. The data use objective for this ME is to assess the adverse effects of chemicals (and evaluation of conventional parameters, such as grain size, total organic carbon [TOC], sulfide, and ammonia) in sediment to the benthic invertebrate community. The results of the toxicity tests are also evaluated using existing background data from similar studies conducted in Jamaica Bay. This LOE is part of the SQT assessment (Appendix A; Section 6).

Risk Question: *Are COPEC concentrations in porewater and/or surface water from the NBSA greater than toxicological benchmarks for the survival, growth, and/or reproduction of invertebrates?*

This question is addressed by comparing COPEC concentrations in porewater and surface water to available toxicological criteria and benchmarks. The data use objective for this ME is to estimate the exposure of the benthic invertebrate community to COPECs in sediment porewater and overlying surface water. This LOE is part of the surface water assessment (Section 5) and SQT assessment (Appendix A; Section 6).

3.1.2 Fish

Assessment Endpoint 2: Survival, growth, and/or reproduction of fish.

The fishery of the NBSA includes species at various trophic levels of the food web. They consume invertebrates and other lower-trophic level organisms but, in turn, are fed upon by upper-trophic level birds and mammals. Similar to invertebrates, there are various taxonomic feeding guilds for fish including benthic-feeding (i.e., benthivores), omnivorous forage fish, and pelagic predatory fish. Each feeding guild has a unique exposure potential to COPECs in NBSA media that is evaluated in this BERA.

Risk Question: *Are fish communities in the NBSA different from those found in similar nearby water bodies?*

This question is being addressed by taking advantage of a substantial spatial and temporal fish community dataset that was generated over more than a decade by the USACE and Port Authority of NY/NJ as part of the NY/~~NJHDP~~NJ HDP. Information from the associated survey reports and literature was used to evaluate the distribution, diversity, abundance, and seasonal patterns of adult fish and ichthyoplankton in the NBSA compared to other areas in the NY/NJ Harbor region. The fish community LOEs are presented in Section 7.

Risk Question: *Are COPEC concentrations in fish tissues from the NBSA greater than tissue (whole-body) toxicity benchmarks for the survival, growth, and/or reproduction of fish?*

Identified fish receptors were collected throughout the NBSA for whole-body chemical analyses. These data are compared to literature-based TRVs and/or to whole-body fish tissue chemical concentrations of selected receptors from background reference locations. Specific species of fish were targeted so that representative species from each of three trophic levels are captured: forage fish, benthic/demersal, and pelagic predatory. Concentrations of COPECs in the tissues (whole-body, liver) of fish from the NBSA were measured, and associated data to help interpret the findings (e.g., percent lipids, fish species, length, weight, age, sex) were collected. Egg concentrations for PCDD/PCDF TEQ and PCBs were also estimated via modeling from existing sediment data. External health observations (gross histological analysis) were conducted on captured fish to provide information on fish population health. The fish tissue LOEs are presented in Section 7.

Risk Question: *Are COPEC concentrations in porewater, surface water, and sediment from the NBSA greater than benchmarks for the survival, growth, and/or reproduction of fish?*

Data on COPECs in porewater and surface water collected from the NBSA are compared to criteria and toxicological benchmarks. Data on COPECs from sediments are used to provide perspectives, relative to the tissue assessment described above, on potential exposure-risks from sediment ingestion. The sediment data are also used to model potential uptake of metals and PAHs into fish to help assess potential risk from chemicals that are taken up, but do not accumulate in the bodies of fish. The data use objective for this ME is to estimate the exposure of fish to COPECs via the surface water, sediment, and sediment porewater exposure pathways. The surface water LOE is presented in Section 5, and the sediment and sediment porewater LOEs are presented in Section 7.

Risk Question: *Does the daily dose of COPECs received by fish (including forage fish, benthic omnivore, pelagic invertivore, and pelagic piscivore) from consumption of the tissues of prey species and from other media in the NBSA exceed the TRVs for survival, growth, and/or reproduction of fish? If yes, what are the probabilities of effects of differing magnitude for survival, growth, and/or reproduction of fish?*

This ME is evaluated by comparing receptor-specific modeled daily doses associated with the ingestion of chemicals in sediment and prey tissue with literature-based dietary dose TRVs. The data use objective for this ME is to estimate exposure of fish receptors via various exposure pathways to chemicals in sediment and prey tissue. Four different feeding guilds are evaluated: forage fish, benthic omnivore, pelagic invertivore, and pelagic piscivore. Surface sediment COPEC data (from the BAZ), and benthic invertebrate and/or fish prey tissue COPEC data, depending on receptor-specific

diet, are used to develop the dietary model for each fish receptor. The fish dietary LOEs are presented in Section 7.

3.1.3 Birds

Assessment Endpoint 3: Survival, growth, and/or reproduction of birds.

Birds are part of the upper trophic levels of the food web. There are three feeding guilds of birds evaluated in this BERA: benthivores (consume benthic invertebrates), omnivores (consume plant and animal matter), and piscivores (consume fish). Each feeding guild has a unique exposure potential to COPECs in NBSA media that is evaluated in this BERA.

Risk Question: *Does the daily dose of COPECs received by birds (including piscivorous, benthivorous/sediment probing, and omnivorous species) from consumption of tissues of prey species and other media in the NBSA exceed toxicity benchmarks for survival, growth, and/or reproduction of birds? If yes, what are the probabilities of effects of differing magnitude for survival, growth and/or reproduction of birds?*

This ME is evaluated by comparing receptor-specific modeled daily doses associated with the ingestion of chemicals in sediment and prey tissue with literature-based dietary dose TRVs. The data use objective for this ME is to estimate exposure of bird receptors via various exposure pathways to chemicals in sediment and prey tissue. Three different feeding guilds are evaluated: piscivorous, benthivorous/sediment-probing, and omnivorous. Surface sediment COPEC data (from the BAZ), and benthic invertebrate and/or fish prey tissue COPEC data, depending on receptor-specific diet, are used to develop the dietary model for each bird receptor.

Risks to birds are also assessed by evaluating field-collected tissue residues of birds (specifically egg tissue) collected from an existing study from Shooters Island (Parsons 2003). The egg tissue data collected from this study are compared to literature-based TRVs for avian eggs, and the study's findings regarding reproductive endpoints are included as direct measures of effects. The bird dietary and egg LOEs are presented in Section 8.

3.1.4 Mammals

Assessment Endpoint 4: Survival, growth, and/or reproduction of mammals.

Mammals are part of the upper trophic levels of the food web. There are two feeding guilds of mammals evaluated in this BERA: omnivores (consume plant and animal matter), and piscivores (consume fish). Each feeding guild has a unique exposure potential to COPECs in NBSA media that is evaluated in this BERA.

Risk Question: *Does the daily dose of COPECs received by mammals (including omnivorous and piscivorous mammals) from consumption of the tissues of prey species and from other media at the NBSA exceed toxicity benchmarks for survival, growth, and/or reproduction of mammals? If yes, what are the probabilities of effects of differing magnitude for survival, growth, and/or reproduction of mammals?*

This ME is evaluated by comparing receptor-specific modeled daily doses associated with the ingestion of COPECs in sediment and prey tissue with literature-based dietary dose TRVs. The data use objective for this ME is to estimate exposure of aquatic and semi-aquatic mammals to COPECs

in NBSA sediment and prey tissue. Two feeding guilds are evaluated: omnivorous and piscivorous. The mammal assessment is presented in Section 9.

3.1.5 Reptiles

Assessment Endpoint 5: Survival, growth, and/or reproduction of reptiles.

Risk Question: *Are COPEC concentrations in site surface water and sediment at levels that might adversely affect survival, growth, and/or reproduction of reptiles?*

This question is primarily addressed qualitatively by discussing the possibility of reptiles inhabiting the NBSA, particularly the diamond-backed terrapin, which is the only species observed or noted in the Bay. In addition, concentrations of COPECs in surface water and surface sediment from subtidal and intertidal areas are compared to available, relevant toxicity-based benchmarks. Finally, the potential risks to reptiles are put into perspective based on a bounding of the results from the bird and mammal assessments, based on the potential relative exposure pathways/magnitude for reptiles compared to these other organisms. The data use objective for this ME is to estimate the exposure of reptiles via direct contact and uptake of COPECs in surface water and sediment. The reptile LOE is presented in Section 10.

3.2 Representative Ecological Receptors

Representative receptor species for the AEs in this BERA were selected based on the available reports, biological surveys, habitat data, and other information from the NBSA, as discussed above. The following factors were considered for the selection of proposed ecological receptors:

- Potential for exposure to sediment-associated chemicals – Ecological receptors exposed to sediments through direct and incidental ingestion of sediment, ingestion of sediment-exposed prey, or direct contact with sediments have the greatest potential for exposure to sediment-associated chemicals, as opposed to species with less direct sediment contact. In addition, ecological species with small home ranges and whose site use is limited to the NBSA (e.g., mummichog) have a greater potential for exposure to site-related chemicals in sediment than do migratory species (e.g., bluefish), species with large home ranges, or species that do not exclusively use aquatic habitats (e.g., raccoon).
- Relative ability to bioaccumulate/biomagnify site-related chemicals – Species from upper-trophic levels (e.g., piscivores) have a greater potential for long-term exposure to bioaccumulative chemicals and a greater potential for the biomagnification of those chemicals.
- Societal and cultural significance (including species that are highly valued by society) – Federally and State-listed threatened and endangered species have special consideration in the selection of receptors. Table 2-20 provides the animal and plant species listed for the NBSA in the states of New Jersey and New York. Also, species that are considered commercially and recreationally important receive greater consideration in the selection of receptors.
- Ecological significance (including species that serve a unique ecological function) – Species with unique foraging preferences, such as those that primarily feed in shallow mudflat areas or are bottom-dwellers, receive special consideration.

- Sensitivity to site-related chemicals – Species with known sensitivities to particular chemicals (e.g., piscivorous birds sensitive to dichlorodiphenyltrichloroethane, mink sensitive to dioxins/furans and PCBs) receive special consideration in the selection of receptors.

Ecological receptors may be directly exposed to chemicals through contact (e.g., direct contact to sediment and/or surface water), through ingestion of chemicals in water or sediments, or indirectly through the ingestion of contaminated food items. For an exposure pathway to be complete, a chemical must be able to travel from the source to receptors that utilize or inhabit the site and be taken up by the receptors via one or more exposure routes.

As described in Section 3.1, the receptor groups that may be exposed to COPECs in sediment, surface water, and/or tissue from the NBSA include benthic invertebrates, fish, birds, mammals, and reptiles. Most of these organisms are subject to being exposed to multiple habitats and areas within the NBSA. Aside from infaunal benthic invertebrates, most species are not sessile and will actively move in and out of different habitats/geomorphic areas, thus exposing them to a wide range of COPECs within NBSA sediments and surface water. For instance, blue crabs inhabit the subtidal flats and intertidal areas in the summer, but will overwinter deep in the navigation channels. Species with known locational (i.e., site) fidelity for all or part of their life history were considered in the selection of receptors. Species were also selected based on feeding guild (e.g., benthic-feeders vs. piscivorous), which reflects differences in exposure. The representative organisms selected to represent each receptor group for the NBSA BERA are listed in Table 3-2. Life history profiles, which summarize the key information regarding the life stages, life cycles, habitat use, and feeding strategies for these receptors, are provided in Appendix B.

Table 3-2. Representative Ecological Receptors

Receptor Group	Receptor Guild and Habitat	Site Use	Species
Benthic Invertebrates	Benthic, deposit/filter-feeders, marine	Resident	Infaunal inverts./ polychaete worms
	Benthic, filter-feeder, marine	Resident	Softshell clam
	Benthic omnivore, marine	Resident	Blue crab
Fish	Forage fish	Resident	Mummichog/killifish (*Fundulus* ~~spp~~sp.)
	Forage fish	Migratory	Atlantic menhaden/silverside
	Benthic omnivore	Migratory	Summer/winter flounder
	Pelagic invertivore/piscivore	Migratory	White perch
	Pelagic piscivore	Part Resident	American eel
Birds	Benthivore/sediment-probing	Migratory	Spotted sandpiper
	Subtidal piscivore	Part Resident	Great blue heron
	Pelagic piscivore	Part Resident	Double-crested cormorant
	Omnivore	Migratory	Lesser Scaup
Mammals	Piscivore	Transient	River otter
	Omnivore	Resident	Muskrat
	Piscivore	Transient	Harbor seal

3.3 Ecological Conceptual Site Model

A preliminary ecological CSM was presented in the Phase I RIWP (Tierra 2005) and in the SLERA (Battelle 2008), and was recently updated in the draft NBSA RI Report (GSH ~~2019, in preparation).~~2020). Based on the review of additional data and reports (i.e., the data and information summarized in this document) and information from the Problem Formulation (Attachment 2; Tierra 2013a) and NBSA Reconnaissance (Tierra 2015a), the ecological/food web components of the CSM have been updated as displayed on Figure 3-2; the specific ecological exposure pathways are depicted on Figure 3-3.

3.3.1 Ecological Exposure Pathways

Following discharge to the NBSA, contaminants can partition between the dissolved- and particulate-phase in the water column. Chemicals in the particulate-phase can then be transferred to the sediment bed by settling or be resuspended from the bed to the water column. The estimated depth of the BAZ, the area where biological activity and benthic exposures to contaminants occur, is approximately 6 inches throughout the Bay (Tierra 2008b). Bioaccumulation of contaminants is dependent on a variety of factors including contaminant type and speciation, metabolism, lipid content of the organisms, and physicochemical attributes of the sediment and porewater (e.g., organic carbon and sulfide content). Contaminants can enter the food web via accumulation in tissues of biota exposed directly to contaminants in the water column, sediments/porewater, and/or in the tissues of prey items (Figure 3-2). Food web interactions are likely the most significant exposure pathways for wildlife that forage in the NBSA. Because the Bay surface water is saline, ingestion of water, other than incidental ingestion during foraging, is not an exposure pathway for wildlife.

For aquatic organisms, such as fish and invertebrates, exposure to contaminated media (i.e., sediment and surface water) occurs via direct contact (e.g., absorption) or ingestion/uptake. Although birds and mammals are in direct contact with contaminated media, exposure via direct or dermal contact is negligible due to the presence of feathers and fur. Field reconnaissance of the NBSA (Tierra 2015a) indicated little evidence of reptile use in the NBSA. Similarly, inhalation of contaminants, such as volatile compounds in sediment or surface water by wildlife, is negligible.

3.3.2 Assessment Zones

Risks under this BERA are estimated both on a NBSA-wide basis, and for a series of assessment zones to evaluate relative risks on most localized areas of potential exposure (Figure 3-4). There are three primary assessment zones for which risks are quantified in this BERA: north, southeast, and southwest. The regularly dredged navigation channel zone is treated qualitatively in this BERA, as only limited data were collected from this zone for risk assessment purposes due to sampling limitations because of the regular ship traffic in the channels, the more temporary nature of the sediments in these channels as a result of regular removal from dredging, and the more limited and transient exposure of fish and other vertebrate organisms in these areas. In addition, the contamination in these areas is evaluated and addressed by the USACE as part of the permitting process for the navigation dredging events. The limited intertidal mudflat habitats that are present in the NBSA are evaluated in this BERA in terms of the specific risk potential from localized sediment contamination to organisms that preferentially forage in these

areas. Shoreline samples, which are also inclusive of all mudflat samples, are evaluated for muskrat only, which preferentially forage in these locations within the NBSA.

The selection of the assessment zones for evaluation were based on geographic considerations related to Bay physiography (i.e., position relative to the navigation channels, representation of different geomorphic features and depths within each non-channel zone, etc.), habitat, and data distribution for the different BERA datasets. The purpose of the zones is to evaluate if more localized exposures, within a reasonable foraging range for the different receptors, would result in estimates of potential exposure risks that differ from those estimated on a NBSA-wide basis. A summary of the zone-specific risk evaluations conducted for each of the key receptors in Sections 6 through 9 is presented in Table 3-3. Discussion of the specific analytical data available for each zone is provided in Section 4.4.

Table 3-3. Summary of Zone-Specific Exposure Risk Analyses for the Representative Receptors

Receptor	Prey Consumption		Incidental Sediment Ingestion			
	NBSA-Wide	Assessment Zone[a]	NBSA-Wide	Assessment Zone[a]	Mudflat	~~Shoreline~~Shoreline[c]
Fish[b]						
Mummichog/killifish	✓	✓			✓	
Summer/Winter Flounder	✓	✓	✓	✓		
White Perch	✓	✓	✓	✓		
American Eel	✓	✓	✓	✓		
Birds						
Spotted Sandpiper	✓	✓			✓	
Great Blue Heron	✓	✓			✓	
Double-crested Cormorant	✓	✓	✓	✓		
Lesser Scaup	✓	✓	✓	✓		
Mammals						
River Otter	✓	✓	✓	✓		
Muskrat	✓	✓				✓
Harbor Seal	✓	✓	✓	✓		

~~Table 3-3~~ **Notes:**
[a] = Preliminary assessment zones (North, Southeast, Southwest); no biota data were collected in ~~Maintained Dredged Channel~~regularly dredged deep water channel zone
[b] = Atlantic menhaden/silverside (migratory forage fish) not included in this table as they are only being assessed from their tissue data
[c] = All intertidal mudflat samples are also considered and included as shoreline samples.
✓ = Risk assessment being conducted for this exposure scenario

3.4 Summary of SLERA and Selection of COPECs

The final SLERA for the NBSA (Battelle 2008) is provided as Attachment 1 to this BERA. The SLERA was conducted using historical datasets that were available for the NBSA prior to implementation of Phase I and II of the RI.

The results of the SLERA indicated that the average and/or upper-bound concentrations of a variety of contaminants in sediments exceeded conservative screening guidelines and/or benchmarks throughout the NBSA. In addition, preliminary screening-level food web model results for fish, birds, and mammals,

indicated that the upper-bound tissue and exposure-modeled concentrations of bioaccumulative contaminants exceeded conservative tissue benchmarks derived from the scientific literature. From the SLERA a preliminary list of COPECs was identified for sediments and the different types of organisms. These included dioxins/furans, metals, PAHs, PCBs, pesticides, and other semivolatile organic compounds (Table 3-4). The conclusion of the SLERA was that a BERA should be performed to evaluate the potential risks from these COPECs.

As part of the Problem Formulation (Attachment 2; Tierra 2013a) development, a revised screening-level assessment was performed using the surface sediment chemistry data collected during the Phase I and II RI investigations. The results of the updated screen confirmed the list of primary chemical groups retained as COPECs for this BERA and helped to identify the most important individual chemicals or compounds in those groups.

For this BERA, all chemical data collected under the Phase III sampling programs (i.e., 2014-2016 datasets) are evaluated with respect to their potential for risk. In the SQT program, all detected analytes in surface sediments (regardless of prior COPEC status) are included in the statistical evaluations to determine if there is a relationship between the analyte and potential effects on the benthic invertebrate community. In the tissue and food web modeling analyses for shellfish, fish, birds, and mammals, a group of widely detected COPECs with known potential for toxicity are evaluated (Table 3-5). Sensitivity analyses are then presented for other analytes (e.g., metals, pesticides) that were less widely detected, and generally have less toxicological potential or information available, in order to assess the likelihood of risk relative to the more prominent COPECs. The latter results are discussed as part of the uncertainty sections for the various receptor groups in Sections 6 through 10 and summarized as part of the risk conclusions in Section 11.

Table 3-5. COPECs Evaluated for Shellfish, Fish, and Wildlife Receptors in the BERA

COPEC	COPECs for the NBSA BERA						
	Benthic Invertebrate Tissue	Fish Tissue	Fish Egg	Fish Diet	Bird Egg	Bird Diet	Mammal Diet
Dioxins/Furans							
2,3,7,8-TCDD	X	X	X		X	X	X
Total Dioxin/Furan TEQ Fish		X	X				
Total Dioxin/Furan TEQ Bird					X	X	
Total Dioxin/Furan TEQ Mammal							X
Polychlorinated Biphenyls							
Total PCBs	X	X	X		X	X	X
Total PCB TEQ Fish		X	X				
Total PCB TEQ Bird					X	X	
Total PCB TEQ Mammal							X
Total Dioxin/Furan/PCB TEQ Fish		X	X				
Total Dioxin/Furan/PCB TEQ Bird					X	X	
Total Dioxin/Furan/PCB TEQ Mammal							X
Pesticides							
Total DDx (2,4 & 4,4)	X	X			X	X	X
Dieldrin	X	X			X	X	X
Total Chlordane	X	X			X	X	X
Hexachlorobenzene	X	X			X	X	X
Polycyclic Aromatic Hydrocarbons							
Total HMW PAH	X					-X	-X
Total LMW PAH	X					-X	-X
Total PAH	~~X~~			X	X	X	X
Inorganics							
Arsenic	X	X		X		X	X
Cadmium	X			X	X	X	X
Chromium	X	X		X		X	X
Copper	X	X		X		X	X
Lead	X	X		X		X	X
Mercury/Methylmercury	X	X	X	X	X	X	X
Nickel	X			X		X	X
Selenium	X	X		X		X	X
Silver	X	X		X		X	X
Zinc	X	X		X		X	X

Notes:
[a] Total chlordane is the sum of heptachlor epoxide, oxychlordane, heptachlor endo-epoxide, trans-nonachlor, alpha-chlordane, cis-nonachlor, gamma-chlordane, and heptachlor.
X = Indicates the constituent is a COPEC for the species and line of evidence

3.5 Risk Analysis Framework for the BERA

Consistent with USEPA guidance (1997, 1998, 1999), risk is evaluated for populations of ecological receptors. Potential effects of COPEC exposure to populations or communities may be measured directly or evaluated using toxicity of individual members of the population based on changes in growth, reproduction, or survival, and extrapolating how these individual effects translate to populations of organisms. The risk analysis framework for this BERA was designed to determine if exposures to COPECs in the NBSA are likely to cause adverse effects to ecological populations at the site or changes in community composition or function.

The risk analysis framework is based on consideration of multiple LOEs, where available, to draw risk conclusions regarding the potential for unacceptable risk to populations of ecological receptors. The various LOE for each receptor group and overall WOE approach used in this BERA are described in more detail below.

3.5.1 Lines of Evidence

According to USEPA (1997) guidance, there are four primary types of LOEs that can be used to determine if ecological risks exist at the site:

1. Comparing estimated or measured exposure levels to levels that are known from the literature to be toxic to receptors associated with the assessment endpoint
2. Comparing the observed effects in the receptors associated with the site with observed effects in similar receptors at a reference site
3. Comparing the results of laboratory bioassays with media from the site and reference sites
4. Comparing the results of *in situ* toxicity tests conducted at the site and reference sites.

For each receptor group evaluated in this BERA, the LOEs with available data are summarized in Table 3-6. Available data are summarized in more detail in Section 4.3.

Table 3-6. Lines of Evidence for the BERA

Receptor Group	Exposure and Toxicity Comparisons (Hazard Quotients)			Site-specific Toxicity Measures	Comparison to Reference (Urban Background)
	Environmental ~~Media[f]~~Media[a]	Dietary	Tissue		
Benthic ~~Invertebrates[a]~~Invertebrates[b]	X		X	X	X
Fish	~~X[b]~~X[c]	X	~~X[e]~~X[d]	~~X[d]~~X[e]	X[f]
Birds		X	X	X[g]	X[g]
Mammals		X			
~~Reptiles[e]~~Reptiles[h]		X			

Table 3-6 Notes:
a. Sediment, porewater, and/or surface water data.
~~a.~~b. Benthic invertebrates were evaluated using the SQT as the primary LOE for infaunal communities, and a tissue assessment for epibenthic shellfish (clam and crab). ~~See Section 6 for more detail.~~
~~b~~c. Surface water data are mostly below available screening levels and do not indicate potential for ecological risk; surface water hazard quotients were not calculated in this BERA. ~~See Section 5.~~
~~c~~d. Whole-body and/or egg tissue.
~~d. Fish~~e. Site-specific fish pathology ~~was evaluated as a supporting LOE~~study.
f. USACE ABS historical adult, juvenile, and ichthyoplankton community studies.
g. ~~e~~Parsons (2003) site-specific bird tissue/egg/reproduction study.
h. Evaluated qualitatively only.
~~f. Sediment, porewater, and/or surface water data.~~

The LOEs align with AEs, risk questions, and MEs as presented in Table 3-1. The various LOEs were used to develop overall risk conclusions based on consideration of the specific strengths and uncertainties for each LOE. For benthic invertebrates, potential for unacceptable risk to benthic invertebrate (infaunal) communities in the NBSA was evaluated using the SQT assessment (Appendix A; Section 6). This approach, which incorporates multiple ~~LOsE~~LOEs (site-specific toxicity testing [including comparison to toxicity in reference locations], measurement of community structure and function, and chemical analysis of environmental media and associated dose modeling), has the most site relevance and least uncertainty in predicting potential for unacceptable risk in the NBSA. For the remaining receptors that are quantitatively evaluated in this BERA (shellfish, fish, birds, and mammals), hazard quotients (HQs) were used as the LOE to define preliminary constituents of concern (COCs); all COPECs with a lowest observed adverse effect level (LOAEL) HQ ≥ 1 in any of the available HQ LOEs (including sensitivity analyses) were identified as a preliminary COC (i.e., having potential for unacceptable risk). Each preliminary COC was then evaluated using ~~other~~the available LOEs (as indicated in Table 3-6) and consideration of the specific strengths and uncertainties for each LOE to draw risk conclusions for that receptor-COPEC pair based on the cumulative WOE. More detail regarding the WOE approach is provided in Section 3.5.3.

3.5.2 Consistency Between the NBSA (OU3) and Lower Passaic River (OU4) BERA

As stated in Section 1, an overarching goal of this NBSA BERA is to maintain, to the extent practicable, consistency with the technical approach and applications for estimating risks and drawing risk conclusions in the LPRSA BERA. While there are differences in the ecosystems between the two sites (i.e., settings, physicochemical characteristics, geomorphology, habitats, data collected, and representative receptor species), in general, the types of receptors and approaches to evaluating risks are similar within the context of the USEPA Superfund ERA guidelines (USEPA 1997).

The risk analysis plan for this BERA is consistent with that of the LPRSA BERA. Important process similarities include the use of an SQT assessment for assessing risk to benthic invertebrate communities, and a tissue and food web exposure modeling approach, along with the calculation of HQs, for epibenthic invertebrates, fish, birds, and mammals.

In order to keep the tissue and food web exposure-risk assessments comparable between the two ~~sites~~OUs, GSH has adopted, to the extent feasible and appropriate, the TRVs for the different COPEC

and receptor groups from the LPRSA BERA (Windward 2019)[3]. This includes the incorporation of the TRVs developed by the USEPA for the *Focused Feasibility Study Report for the Lower Eight Miles of the Lower Passaic River* (LPR FFS; USEPA 2014) for risk bounding purposes.

In general, regarding bounding of risk estimates, the parties concur that the LPR FFS TRVs are more conservative than comparable TRVs developed by the CPG for the LPRSA BERA. The LPRSA BERA TRVs used in risk calculations were selected from a larger dataset of no observed adverse effect level (NOAEL) and LOAELs in one of two ways: 1) a TRV (representing a LOAEL TRV) was derived as the 5th percentile value of a species sensitivity distribution (SSD) curve if toxicity data were sufficient and the highest NOAEL below the 5th percentile SSD value was selected as the NOAEL, or 2) a study-specific NOAEL TRV and a LOAEL TRV were selected (if toxicity data were not sufficient to develop an SSD). When data were not sufficient to derive a TRV from an SSD, the lowest acceptable LOAEL and the highest NOAEL below the LOAEL from the same study were selected as the TRVs. The SSD and TRVs development and selection process for the TRVs used in the LPRSA BERA are described in detail in the main text and Appendix E of that document.

The intent of using both TRV sources, when available, is to understand the potential range of effects within a particular trophic category from more sensitive to less sensitive endpoint receptors. Different TRVs for the NBSA BERA were only selected ~~when available toxicological data was available for marine organisms~~for COPECs that were ~~more comparable to those~~not assessed in the ~~Bay, versus freshwater organisms assessed in the LPR~~LPRSA BERA. By applying common TRVs to both BERAs, the risk estimates are analogous between the two BERAs. This BERA cites the CPG's LPRSA BERA (Windward 2019) for details on the process in developing the common set of TRVs for invertebrates, fish, birds, and mammals, and does not repeat this detailed information. Summary information on TRV development and selection is provided in the respective sections and appendices of this BERA. Toxicity profiles and discussions on the TRVs are presented in detail in Appendix D.

The interpretation of the risk calculations/estimates for this BERA is also consistent with the LPRSA BERA. In the risk characterization, LOAEL HQs ≥ 1 are used as the initial threshold for potential unacceptable risk and evaluation of LOEs for each receptor are considered in a WOE approach. The outcome of the WOE approach is a list of, and basis for, recommended COCs for further evaluation in the FS. The approach is described in more detail in Section 3.5.3.

3.5.3 Weight of Evidence Approach for Drawing Risk Conclusions

At the conclusion of this BERA, COPECs with uncertain or unacceptable risk to ecological receptors (i.e., risk drivers) are identified and these constituents are recommended as final COCs for further evaluation in the FS. To identify these risk drivers/final COCs, a WOE approach was used. In the WOE evaluation, the various LOE available for each receptor group and COPEC are evaluated for factors contributing to the overall confidence and conservatism inherent in the risk estimates. Through consideration of the

[3] It is the NJDEP's position that a single TRV set (NOAEL and LOAEL) that evaluates the more sensitive species and endpoints to characterize risk to invertebrates, fish, birds and wildlife should be selected in a BERA, not two sets of TRVs as presented in this document. The NJDEP's Ecological Evaluation Technical Guidance, August 2018, does not advocate the use of more than one set of TRVs for individual contaminant-receptor pairs. It is the NJDEP's position that use of one conservative TRV set derived for sensitive receptors and sensitive endpoints most clearly demonstrates the degree of risk for individual contaminant-receptor pairs and ensures protection of threatened, endangered and species of special concern.

independent strengths and weaknesses of each LOE, it is possible to draw risk conclusions supported by the cumulative WOE.

As presented by Suter (2006), factors to consider when evaluating various LOE are related to the overall relationship of the LOE to the AE, the quality of the data, and the strength of the study design to address risk questions at the site. These factors can include site relevance, existence of a dose-response relationship, spatial extent, quality and quantity of the data, which are discussed briefly below:

- Site Relevance – LOE and supporting data (such as toxicity values or exposure factors) derived from information with high site relevance was given higher weight than information with unknown or unlikely relevance to the NBSA. Representativeness of the media, species, environmental conditions, and habitat type relative to those at the site is considered. For example, toxicity data derived for species present at the site or derived using realistic exposure routes are given higher weight than for less relevant data.
- Dose-Response Relationship – The existence of a dose–response relationship and the strength of the relationship increase confidence in the LOE.
- Spatial and Temporal Extent – A LOE was given more weight if the data adequately represent the area assessed. For example, tissue data collected from resident species are given higher weight than for data collected from migratory species with other potential non-site exposures; exposure estimates representative of a receptor population are preferred over estimates based on much smaller or larger areas.
- Quality and Quantity - Various factors, including analysis procedures, sample counts, and frequency of detection (FOD), can be considered to assess the LOE. In general, data with adequate ~~detection~~quantitation limits, high relative sample counts, and low FOD were given higher weight.

For example, when interpreting HQ results, uncertainties in the various calculation inputs are considered. These can include supporting statistical information for the exposure estimates (e.g., sample size, FOD), confidence in the toxicity values and exposure factors, the direction of uncertainty in the risk estimates, and spatial extent of elevated concentrations.

Potential for unacceptable risk to benthic invertebrate (infaunal) communities in the NBSA was evaluated using the SQT assessment (Appendix A; Section 6). This approach, which incorporates multiple LOE (site-specific toxicity testing, measurement of community structure and function, and chemical analysis of environmental media), has the most site relevance and least uncertainty in predicting potential for unacceptable risk in the NBSA. The SQT data are site-specific and designed to evaluate the various habitat types and invertebrate communities present within the NBSA. The robust statistical evaluations incorporated into the SQT integrate exposure from all routes.

For the remaining receptors that are quantitatively evaluated in this BERA (shellfish, fish, birds, and mammals), all COPECs with a LOAEL HQ ≥ 1 in any of the available HQ LOEs (including sensitivity analyses) were identified as a preliminary COC (i.e., having potentially unacceptable risk). Each preliminary COC was then evaluated using all available LOEs and consideration of the factors above to reach a risk conclusion for that receptor-COPEC pair based on the cumulative WOE. Only receptor-COPEC pairs with a LOAEL HQ ≥ 1 and factors supporting the conclusion of unacceptable risk were recommended as risk drivers/final COCs. In cases where no risk conclusion could be made, risk was

considered uncertain and these COPECs were also recommended as final COCs. This approach is consistent with that used in the LPRSA BERA (Windward 2019) and was retained for this BERA for consistency.

There are ultimately three possible risk outcomes for fish, shellfish and wildlife based on the LOAEL HQs ≥ 1 and WOE:

1. Unacceptable risk is possible (i.e., indicated by sufficient and strong supporting LOEs). These COPECs are recommended as final COCs.
2. Unacceptable risk is unlikely (i.e., indicated by sufficient and strong LOEs supporting a conclusion of no unacceptable risk). These COPECs are not recommended as final COCs.
3. Unacceptable risk is uncertain (i.e., indicated by insufficient LOEs). While sufficient data were collected to evaluate ecological receptor exposures in the NBSA, toxicity data used to evaluate the exposures were selected primarily from published literature. Toxicity data for some receptors, such as tissue copper effects levels in fish, are limited in the literature. In cases where there is low confidence in the toxicity data and the various LOEs gave conflicting results, the available information was considered too uncertain to draw risk conclusions. These COPECs are recommended as final COCs.

Selection of final COCs for evaluation in the FS will be completed in consultation with USEPA and based on the findings of this BERA.

The risk characterization results based on the WOE are presented in Section 6.5 for shellfish (i.e., clam and crab), Section 7.7 for fish, Section 8.4 for birds, and Section 9.3 for mammals. A summary of the overall risk findings for this BERA is presented in Section 11.

4 DATA RESULTS AND EVALUATION

This section describes the datasets used to conduct this BERA. A summary of the site-specific investigations performed in Phase III of the RI/FS to support this BERA is provided, along with historical datasets that are used throughout this BERA to provide perspectives on the ecosystem and risk estimates.

4.1 Data Used in the Baseline Ecological Risk Assessment

A Phase III site investigation (SI) and several additional sampling programs to collect the data necessary to conduct the BHHRA and BERA, were conducted between 2012 and 2016. Field activities for the Phase III SI were conducted in 2016 and were performed consistent with the Phase III QAPP (Tierra 2016). Under the RI, two bathymetry surveys (2005 and 2012/2013), an evaluation of the BAZ, and an evaluation of sediment data and sedimentation rates (Tierra 2008a, 2008b, 2011, 2014c) were performed.

In addition, a shoreline/habitat survey and three sampling programs were conducted as part of the Phase III field activities to support the BERA and BHHRA. A shoreline/habitat survey was conducted in 2013 in accordance with the Reconnaissance Survey Work Plan (Tierra 2013c), a crab and clam sampling program was conducted in 2014 in accordance with the Crab and Clam QAPP (Tierra 2014a), a fish sampling program was conducted from 2014 through 2016 in accordance with the Fish Sampling and Analysis QAPP (Tierra 2014b), and a SQT and porewater sampling program was conducted in 2015 in accordance with the SQT and Porewater QAPP (Tierra 2015b).

Two prior Phase I/II investigations were also considered in this BERA. First, an evaluation of the BAZ was conducted in 2005 to evaluate the level of bioturbation that plays a key role in distributing pollutants in sediments (Tierra 2008b). From this work, the depth of the BAZ was defined to be approximately 0 to 6 inches, information that was used to guide the design of the Phase III and risk assessment surface sediment investigations. Second, surface water sampling conducted by the CPG within the NBSA (2010 to 2013) as part of fate and transport modeling work being conducted for the system was evaluated to characterize the potential exposures of ecological receptors in the Bay to contaminants in surface water.

Additional information regarding these field activities are provided in the following reports:

- Estimation of the Biologically Active Zone (Tierra 2008b)
- Newark Bay Study Area Reconnaissance Survey Report (Tierra 2015a)
- Technical Memorandum: Risk Assessment Field Sampling and Analysis Program - Newark Bay Study Area (Arcadis 2015)
- SQT and Porewater Field and Data Reports (GSH 2017a,d)
- Phase III Sediment Investigation Field Report (GSH 2017c)
- Crab and Clam Field and Data Reports (Normandeau 2017; GSH 2017b)
- Fish Field and Data Reports (Normandeau 2017; Tierra 2017).

A summary of the overall components of the data collected under the NBSA Phases I through III field activities used for risk assessment is provided in Tables 4-1a and 4-1b. The list of chemical constituents analyzed for in water, sediment and tissue samples is provided in Table 4-2.

Maps of the sampling locations for the risk assessment program are provided on Figures 4-1 through 4-3 (surface water, SQT, and Phase III sediment sampling locations, respectively), Figure 2-5 (clam/crab sampling locations) and Figures 2-8 through 2-10 (2014-2016 fish sampling locations). Figure 3-4 shows the aggregate BERA sampling locations in the five BERA assessment zones.

Together, these investigations provide the necessary data to complete this BERA, as discussed in the CSM (GSH ~~2019~~2020). The Phase III RI/BERA investigations utilized to conduct this BERA are discussed in sequence of performance in Section 4.3, and the results for the full suite of chemical analytes are presented in that section. A more focused evaluation and summary of the data for the COPECs (identified in Section 3.4) is presented in Section 4.4. This includes the development of the sediment and tissue data inputs, both Bay-wide and for each BERA assessment zone, used in the exposure modeling and risk calculations in Sections 6 through 9.

4.2 Data Reduction Rules

This section describes data reduction and handling methods used to prepare the laboratory analytical data for detailed evaluation (e.g., field duplicates, non-detects) and various data normalization techniques. In addition, the statistical and data visualization methods used to evaluate the various datasets for this BERA are described. Sample results that were rejected in the data validation process were not included in this BERA.

4.2.1 Field Duplicates

Field duplicate sample results were combined with the parent sample (i.e., corresponding sample) results to generate a single sample result for subsequent data evaluation.

1. When both field duplicate and parent sample results were reported as detected concentrations, the sample concentrations were averaged to generate a single concentration.
2. When either the field duplicate sample or the parent sample was reported as a detected concentration, but was not detected in the other sample, only the detected concentration was used to represent the concentration of that analyte in that sample. The alternate approach of averaging the detected value with the ~~detection~~quantitation limit from the non-detect value would introduce unnecessary uncertainty because the ~~detection~~quantitation limit does not provide a known measure of concentration.
3. When a given analyte was not detected in either the parent or field duplicate sample the lower of the two ~~detection~~quantitation limits was used to represent the ~~detection~~quantitation limit for that analyte in that sample.

4.2.2 Calculated Totals

For several chemical groups, data were evaluated both as individual chemicals and as total concentrations for a particular chemical group. Total concentrations (other than those reported by the analytical laboratory[4]) were defined as the sum of the detected concentrations of individual chemicals within that chemical group; a value of zero is used for non-detects. These total concentrations were calculated for the following groups:

1. Total Low Molecular Weight (LMW) PAHs = sum of naphthalene, 2-methylnaphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, and anthracene
2. Total High Molecular Weight (HMW) PAHs = sum of fluoranthene, pyrene, benzo(a)anthracene, chrysene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, indeno(1,2,3-cd)pyrene, dibenzo(a,h)anthracene, and benzo(g,h,i)perylene
3. Total PAHs = sum of naphthalene, 2-methylnaphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benzo(a)anthracene, chrysene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, indeno(1,2,3-cd)pyrene, dibenzo(a,h)anthracene, and benzo(g,h,i)perylene
4. Total Alkylated PAHs = sum of C1-Chrysenes, C1-Fluoranthenes/Pyrenes, C1-Fluorenes, C1-Naphthalenes, C1-Phenanthrenes/Anthracenes, C2-Chrysenes, C2-Fluoranthenes/Pyrenes, C2-Fluorenes, C2-Naphthalenes, C2-Phenanthrenes/Anthracenes, C3-Chrysenes, C3-Fluoranthenes/Pyrenes, C3-Fluorenes, C3-Naphthalenes, C3-Phenanthrenes/Anthracenes, C4-Chrysenes, C4-Naphthalenes, and C4-Phenanthrenes/Anthracenes
5. Total Aroclor PCBs = sum of Aroclor-1016, Aroclor-1221, Aroclor-1232, Aroclor-1242, Aroclor-1248, Aroclor-1254, Aroclor-1260, Aroclor-1262, and Aroclor-1268
6. Total PCBs (209 Congeners) = sum of 209 individual congener PCBs (note that co-elutions were included at the full value for this calculation)
7. Total Alpha + Gamma Chlordane = sum of alpha-Chlordane and gamma-Chlordane
8. Total Chlordane = sum of alpha-Chlordane and gamma-Chlordane, heptachlor epoxide, oxychlordane, heptachlor endo-epoxide, trans-nonachlor, and cis-nonachlor
9. Total BHC Compounds = sum of alpha-BHC, beta-BHC, delta-BHC, and gamma-BHC (lindane)
10. Total DDx (2,4') = sum of 2,4'- dichlorodiphenyldichloroethylene (DDE), 2,4'-dichlorodiphenyldichloroethane (DDD), and 2,4'- dichlorodiphenyltrichloroethane (DDT)
11. Total DDx (4,4') = sum of 4,4'-DDE, 4,4'-DDD, and 4,4'-DDT
12. Total DDx (2,4' and 4,4') = sum of 2,4'-DDE, 2,4'-DDD, 2,4'-DDT, 4,4'-DDE, 4,4'-DDD, and 4,4'-DDT

If all analytes that made up a given total for a specific sample were non-detect, the total for that sample was reported as a non-detect at the highest ~~reporting~~quantitation limit of the individual analytes.

[4] The total concentration of each of the eight dioxin and furan homologue groups was reported by the laboratory.

For uncertainty and sensitivity analyses, totals concentrations were evaluated with either one-half the ~~reporting~~quantitation limit substituted for non-detects in the summation (denoted as ND = ½) or with the full ~~reporting~~quantitation limit substituted for non-detects in the summation (denoted as ND =1).

4.2.3 TEQ Methodology

The toxic equivalency (TEQ) is a toxicity-weighted value based on the estimated toxicity of various dioxin-like compounds relative to 2,3,7,8-TCDD. Each compound is associated with a toxic equivalency factor (TEF) that represents its toxicity relative to 2,3,7,8-TCDD. The TEQ is calculated as the sum of the concentrations of the dioxin-like compounds multiplied by their TEFs. TEFs have been separately derived for fish, birds and mammals (USEPA 2008). The fish and bird TEFs are tissue based while the mammal TEFs are dietary based (USEPA 2008). Three separate totals were calculated for each species group: total dioxin TEQ (based on 2,3,7,8-substituted dioxin and furan congeners only); total PCB TEQ (based on dioxin-like PCB congeners only) and total dioxin/PCB TEQ (based on all dioxin-like compounds). The toxicity-weighted totals for PCBs and PCDDs/PCDFs were calculated by summing each of the detected chemical constituents multiplied by its respective TEF.

There are a number of uncertainties associated with this methodology, as discussed in USEPA (2008); these uncertainties are discussed in the context of receptor group-specific risk characterization in Sections 6 through 9:

- A number of relative potencies for each dioxin-like compound, each derived from its own study, were used to derive a consensus value for the TEF. These relative potencies may vary because of uncertainties in the various steps leading to the determination of value in each study. Such uncertainties include differences in study design and calculation techniques, measurement errors, precision of dose and effects measurements, and natural variability among organisms of the same species in their responses (USEPA 2008).
- The TEFs are point estimates derived from the individual relative potency studies, and they may range over several orders of magnitude among species within each of the groups (i.e., fish, birds, and mammals). There is uncertainty associated with the method used to aggregate the data used to derive each TEF (USEPA 2008).
- The TEQ approach assumes that the toxicity of each dioxin-like compound is additive. It is possible that synergistic or antagonistic interactions could occur.
- The TEFs used in this BERA include only the PCBs, PCDDs, and PCDF congeners known to elicit responses mediated by the aryl hydrocarbon (Ah) receptor.

In addition, recent studies have found that other congeners are more toxic than 2,3,7,8-TCDD, and that the current TEF of 1.0 for two PCDFs (2,3,4,7,8-pentachlorodibenzofuran and 2,3,7,8-tetrachlorodibenzofuran) may underestimate avian toxicity TEFs (Farmahin et al. 2012; Cohen-Barnhouse et al. 2011; Yang et al. 2010). Despite some inherent uncertainties, the TEQ methodology provides a reasonable, scientifically justifiable, and widely accepted method for estimating risks to ecological receptors in CERCLA risk assessments (USEPA 2008).

For uncertainty and sensitivity analyses, total TEQ concentrations were also evaluated with either one-half the ~~reporting~~quantitation limit multiplied by the corresponding TEF substituted for non-detects in the

summation (denoted as ND = ½) or with the full ~~reporting~~quantitation limit multiplied by the TEF substituted for non-detects in the summation (denoted as ND =1).

4.2.4 Normalization

4.2.4.1 Sediment Data Normalization

The spatial and temporal distribution of contaminants in sediments can be affected by interactions between contaminants and grain size, mineralogy, and organic carbon. Varying sediment sources and flow velocities can affect the size distribution and characteristics of materials deposited in specific portions of the bay. These hydrodynamic-related variations in the sediment characteristics can in turn influence the concentrations of contaminants accumulated in the sediment. Most natural and anthropogenic substances, including metals and hydrophobic organic compounds, have a higher affinity to fine-grained particulates than to the coarse fraction. This affinity is related to the surface area of the particles because fine-grained particles have more surface area for adsorption. Often organic matter and clay minerals, associated with fine particle fractions, show the strongest sorption capacity for contaminants. Therefore, even if overlying water concentrations of the contaminants are held constant, there will be a correlation of the contaminant concentration with sediment organic carbon content, clay fraction, or other geochemical factors. The challenge is to separate the natural variability component of sediment mineralogy from the anthropogenic contribution of contaminants to sediment.

Normalization is a data analysis procedure that takes into account both background and contaminant concentrations for the influence of the natural variability in sediment size, mineralogy, and organic carbon content driven largely by differences in energy in the aquatic system (Kersten and Smedes 2002). Through normalization to a parameter that reflects naturally occurring changes in contaminant concentrations, the inherent variability relating to that parameter can be reduced, allowing more meaningful assessment of spatial and temporal patterns in contaminant concentrations.

For hydrophobic organic compounds, concentrations are usually normalized to the organic carbon content of the sediments.

For metals, there is no consensus on the most appropriate constituent to be used in the normalization. Aluminum, iron, TOC, and grain size often vary with sediment texture and show strong relations with metals and can be used as geochemical normalizers (Horowitz 1991; Naval Facilities Engineering Command 2003; San Francisco Estuary Institute 2006). In these cases, the metal of concern may vary with natural differences in sediment composition and mineralogy, but will vary in a relatively constant proportion to the parameter used for the normalization.

A detailed evaluation of normalization for NBSA sediment data is provided in Section 4 (Nature and Extent of Contamination) of the draft RI Report (GSH ~~2019, in preparation).~~2020). The results indicate there was no relationship between inorganic contaminants and normalizing factors. However, nonionic organic contaminants do tend to covary with organic carbon content in the sediments. For the purposes of this BERA, organic COPECs in surface sediments are normalized to TOC, and these results are evaluated in terms of whether they substantially explain variability in the dry weight data results. This evaluation is provided in Section 4.3.4.

4.2.4.2 Biota Tissue Normalization

Lipid levels vary seasonally in organisms (depending on reproductive state) and can also be influenced by a number of environmental factors, including quality of food and stress levels in the organism. Lipid levels in softshell clam were relatively low (mean= 0.36 %; range 0.2 -1%). %) compared to those reported for clams sampled in studies from other Atlantic estuarine areas (e.g., 5.3 - 9% in Boston Harbor; Lohmann et al. 2004). Biota contaminant concentration is normalized to lipid content to potentially minimize the variability associated with changes in lipid content and facilitate comparisons across locations and species. However, lipid normalization is only useful when a direct and proportional relationship exists between lipid content and contaminant concentrations. The relationship between lipid content and contaminant concentration by organisms sampled is explored in Section 4.3.7.

4.2.5 Significant Figures

Tracking of significant figures is important when calculating averages and performing other data summaries. The appropriate number of significant figures associated with specific risk estimates was applied in the last step of each calculation and reflected the least precise value in the calculation (i.e., the lowest number of significant figures).

4.2.6 Calculating Upper Confidence Limits

Exposure point concentrations (EPCs) were represented by the 95% upper confidence limit (UCL) of the mean and were calculated using the USEPA's ProUCL® statistical package (Version 5.1.00) (USEPA 2015b) and were derived following the USEPA guidance for calculating UCLs for EPCs at hazardous waste sites (USEPA 2002). The 95% UCL recommended by the USEPA's ProUCL® software was used. For some data sets, ProUCL may recommend multiple values for the 95% UCL which are all considered valid and in general, multiple recommended values are very similar. If multiple UCLs were recommended by ProUCL, the lowest value was selected to represent the EPC. The UCL is a statistic that estimates the mean concentration with a specified degree of confidence, and accounts for variability among the sampling data. For datasets with fewer than six detected samples, a UCL was not calculated, and instead the maximum concentration was used to represent an EPC. In cases where statistically derived UCLs were greater than the maximum detected concentration, the maximum detected concentration was used in place of the UCL concentration to represent the upper bound value. The 95% UCLs used to represent EPCs in this BERA are summarized in Appendix C.

4.2.7 Treatment of Non-Detects in Risk Calculations

Historically, it has been common practice to estimate the concentration of a chemical not detected in a sample by assuming the chemical is present at a concentration below the detectionquantitation limit. This approach is typically referred to as a substitution method whereby the detectionquantitation limit, one-half the detectionquantitation limit, or a value of zero is used as an estimate of the concentration. In this BERA, the detection limit is defined as the reporting limit and for all analytes the reporting limit is the quantitation limit for the analyte and analytical method.methods, non-detect values were reported to the quantitation limit.

For the purposes of calculating summary statistics (e.g., calculation of mean and standard deviation), one-half the ~~reporting~~quantitation limit was substituted for non-detects. However, when all data are reported as non-detect, the calculated mean and standard deviation are calculated using the full ~~reporting~~quantitation limit; a “less than” symbol (<) is used to indicate a calculated value below the maximum ~~reporting~~quantitation limit.

For the purposes of data presentation in scatter plots and box plots, individual non-detect sample results are shown at the ~~reporting~~quantitation limit~~.~~ and identified as non-detects by symbol or color-coding. Arithmetic means shown on box plots are calculated with one-half the ~~reporting~~quantitation limit substituted for non-detects.

Only detected concentrations were included in totals concentrations as described in Sections 4.2.2 and 4.2.3. However, uncertainty analyses include evaluation of substituting both one-half and the full ~~reporting~~quantitation limit for non-detects in summations.

ProUCL® uses Kaplan-Meier methods for handling non-detect data (USEPA 2015b) in the calculation of UCLs. Input data sets for ProUCL identify non-detects and include the ~~reporting~~quantitation limit.

4.2.8 Graphical and Statistical Evaluations

Boxplots are used to graphically display measurements in a sample population for the analyses included in Section 4 of this BERA. Boxplots display the main distributional characteristics (spread and skew) of the data. The "box" contains 50% of the sample values from the 1st quartile (25th percentile) to the 3rd quartile (75th percentile), also referred to as the inter-quartile range. The sample median (50th percentile) is displayed as horizontal line within the box. The whiskers represent the smallest or largest value within 1.5 times the interquartile range above or below the box. Mild outliers are shown as individual points that lie outside 1.5 times the interquartile range above or below the box. Extreme outliers are shown as individual points that lie outside 3 times the interquartile range above or below the box. Non-detect data are plotted at the ~~reporting~~quantitation limit. Percentiles are based on the full ~~reporting~~quantitation limit for non-detects. The arithmetic mean shown on box plots is calculated similarly as the summary statistics presented in tables with one-half the ~~detection~~quantitation limit substituted for non-detects. Means and percentiles are not shown on box plot graphics when sample size is less than 8 or FOD is less than 50%.

In addition to the graphical displays, several statistical evaluations were conducted on the data as part of the BERA to glean information on the similarity or difference in contaminant concentrations between the various spatial features or areas of the NBSA. These tests included:

> Analysis of variance (ANOVA): ANOVA procedures were used to evaluate whether significant differences exist in mean contaminant concentrations. ANOVA is a linear model in which the dependent variable is continuous and there is one independent fixed effect categorical variable with two or more possible values. One-half the ~~detection~~quantitation limit was substituted for non-detects. Data were log-transformed to better satisfy the assumptions of normality of data residuals and homogeneity of variance. Residuals were checked for approximate normality and symmetry (Scheffe 1959). In addition, within group standard deviations were checked for departure from homogeneity as described by Hochberg and Tamhane (1987). When ANOVA models were statistically significant ($\alpha < 0.05$); a post-hoc Tukey-Kramer pairwise multiple comparison test (Kramer 1956) was conducted to determine which group or groups within the categorical variable set were different from

each other. Log-transformation results in the comparison of geometric means; however, under the assumption of equal variance, differences in means are inferred when differences in geometric mean are statistically significant.

Spearman Rank Correlation: The strength of association between toxicity and benthic effects measures and contaminant concentrations was evaluated using Spearman's rank-order correlation. The Spearman's rank-order correlation is the nonparametric version of the Pearson product-moment correlation. Spearman's correlation measures the strength and direction of a monotonic association between two variables. When one variable generally increases (or decreases) as the other variable increases (or decreases), the two variables are said to possess a monotonic correlation. Correlation coefficients have the characteristic of being dimensionless and scaled to lie in the range −1 to 1. When there is no correlation between two variables, the correlation coefficient is equal to zero (0). When one variable increases as the second increases, the correlation coefficient is positive. Alternatively, when one variable decreases as the second increases, the correlation coefficient is negative.

4.3 Phase III RI/BERA Data Collections and Results

The following sections discuss the results of the data collected for use in this BERA. Summary statistics are provided for the full suite of detected analytes (including, but not limited to the focused list of COPECs) in this section. More focused data summaries for the COPECs are provided in the respective risk assessment sections of this BERA. Appropriate normalizations and data reductions are also presented and discussed in this section, along with broad spatial findings from the various datasets. Detailed discussions of the data findings for the NBSA are provided in the draft RI Report (GSH ~~2019, in preparation~~2020).

4.3.1 Shoreline/Habitat Survey

The Shoreline/Habitat Reconnaissance Survey for the NBSA was conducted in 2013 as part of the BHHRA and BERA activities for the RI/FS. The survey was completed by boat and video, and still cameras were used to record various shoreline/habitat data, including:

- Shoreline/habitat characterization information
- Characterization of human use of shoreline areas
- Identification of sampling areas and evaluation of sampling approaches.

Where feasible, the field biologists surveyed, on foot, the shoreline areas identified as habitat looking for bird nests and signs of mammal use (e.g., scat, tracks) and documented such findings. Survey findings, including a photograph log and a shoreline characterization video, are provided in the Reconnaissance Survey Report (Attachment 3; Tierra 2015a). Figure 2-11 shows the shoreline habitat characterization from Figure 2-1 overlain with evidence of bird or mammal use that were documented by the field biologists during the 2013 Reconnaissance Survey.

The findings of the reconnaissance were used to help design the various sampling programs in Phase III of the RI/FS, particularly the biota sampling locations in nearshore (intertidal and shallow subtidal) areas. The results from the survey were used to develop the shoreline, habitat, and species characterizations

presented in Section 2, and are discussed in various portions of the risk analyses in Sections 6 through 10.

4.3.2 Surface Water Chemistry Data

A comprehensive surface water sampling program was conducted by the CPG within the LPR and Newark Bay (2010 to 2013) as part of the fate and transport modeling work being conducted by the CPG for the LPRSA and NBSA (AECOM 2010, 2011). These surface water data have been deemed sufficient by the USEPA for use in conducting the RI, BHHRA, and BERA for the NBSA (Arcadis 2015).

Surface water samples were analysed for the parameter groups listed in Table 4-2. The sampling program included six stations (NNE, NBN, NNW, NBE, NBS, and ARK) within the NBSA (Figure 4-1). Other locations outside the NBSA that were also sampled under this program included stations within the Passaic River and its tributaries, the Hackensack River, and the Kill van Kull.

The physical water column monitoring program was conducted from March through November 2010.

Five routine chemical water column monitoring events were conducted in 2011 and 2012 under normal flow conditions (i.e., 400 to 3,000 cubic feet per second [cfs] at Dundee Dam):

- Event 1 was conducted from August 15 to 17, 2011, during average tide (median flow at Dundee Dam was 2,650 cfs)
- Event 2 was conducted from February 20 to 21, 2012, during spring tide (median flow at Dundee Dam was 699 cfs)
- Event 3 was conducted from March 26 to 27, 2012, during neap tide (median flow at Dundee Dam was 392 cfs)
- Event 4 was conducted from June 4 to 5, 2012, during spring tide (median flow at Dundee Dam was 1,389 cfs)
- Event 5 was conducted from December 10 to 11, 2012, during average tide (median flow at Dundee Dam was 664 cfs).

A single low-flow (i.e., less than 400 cfs at Dundee Dam) water column monitoring event was conducted in August 2012 during spring tide (median flow at Dundee Dam was 253 cfs). No samples were collected from the NBSA during this event per the Quality Assurance Project Plan/Field Sampling Plan Addendum. Remedial Investigation Water Column Monitoring/Small Volume Chemical Data Collection (AECOM 2011) because interaction between the lower Passaic River and the NBSA is diminished during low flow events.

Two high-flow events (i.e., greater than 3,000 cfs at Dundee Dam) were conducted in 2013:

- Event 1 was conducted from February to March 2013
- Event 2 was conducted in June 2013.

At each location and event, grab samples were collected from two depths in the water column: 3 feet (0.9 meter) above the bottom and 3 feet (0.9 meter) below the surface. Samples were collected at four intervals (i.e., high water slack tide, low water slack tide, maximum ebb tide, and maximum flood tide) at each location and depth.

Results of the surface water sampling program are summarized in Table 4-3. In general, the concentrations of COPECs were consistent across sampling locations, depths, tidal cycles, and season. The results of the surface water program and perspectives on the findings with respect to the exposure risks for the concentrations of COPECs detected are discussed in Section 5.

4.3.3 Sediment Quality Triad and Porewater Chemistry Data

A SQT and porewater sampling program was conducted in September 2015. The SQT and porewater program was conducted in support of the BHHRA and BERA and in accordance with the SQT and Porewater QAPP (Tierra 2015b). This program was implemented to meet the following data use objectives:

- Evaluate whether exposure to site-related COPECs in the NBSA poses unacceptable risks to benthic invertebrate communities
- Evaluate whether the consumption of benthic invertebrates (represented by a laboratory- exposed bioaccumulation test) poses unacceptable risks to fish and wildlife receptors
- Evaluate whether exposure to site-related COPCs in surface sediments in the shoreline intertidal Areas of the NBSA poses unacceptable risks to human receptors.

Sediment grab samples were collected from 43 locations over 13 days. The SQT analyses (i.e., sediment chemistry, toxicity testing, and benthic invertebrate community analysis) and porewater chemistry analyses were conducted at 30 of these locations. Samples from the remaining 13 locations were evaluated only for sediment chemistry for the purposes of quantifying contaminant levels in shoreline/nearshore areas where potential human exposure by direct contact may occur (Figure 2-5). Sediment samples were analyzed for the parameter groups listed in Table 4-2.

Details of the field program are provided in the *Sediment Quality Triad and Porewater Field Report* (GSH 2017c). Results for the sediment quality triad and porewater sampling program are provided in the *Sediment Quality Triad and Porewater Data Report* (GSH 2017a). Field, laboratory, data quality assurance/quality control (QA/QC) procedures, and data validation were also implemented, as specified in the SQT and Porewater QAPP (Tierra 2015b).

The respective component datasets from the SQT program are summarized in subsections below, consistent with the type of data being described (e.g., sediment, toxicity, porewater).

4.3.4 Sediment Chemistry Data

Under the SQT program, surface sediment grab samples were collected from 43 locations over 13 days. The SQT analyses (i.e., sediment chemistry, toxicity testing, and benthic invertebrate community analysis) and porewater chemistry analyses were conducted at 30 of these locations. Samples from the remaining 13 locations were evaluated only for sediment chemistry for the purposes of quantifying contaminant levels in shoreline/nearshore areas where potential human exposure may occur (Figure 2-1). Sediment samples were analyzed for the parameter groups listed in Table 4-2.

The Phase III sediment investigation program was conducted in fall 2016, pursuant to the Phase III QAPP (Tierra 2016). This program was implemented to provide a current surface sediment chemistry dataset for the entire NBSA based on a statistically developed grid design to meet the characterization needs for the

RI and risk assessments, and provide a basis for future monitoring events to conduct trends analyses. Surface sediment samples were collected using vibracore and PONAR grab samplers. A total of 173 discrete vibracore surface sediment samples and 47 grab samples were collected in the NBSA under this program (Figure 4-3). The vibracore samples and 10 of the grab samples were analyzed as discrete samples, while the remaining grab samples were combined into 10 composite samples. Sediment samples were analyzed for the parameter groups listed in Table 4-2.

Details of the field program and analytical results are provided in the *Phase III Sediment Investigation Field Report* (GSH 2017d) and in the Draft RI Report (GSH ~~2019~~2020). Field, laboratory, data QA/QC procedures, and data validation were also implemented, as specified in the Phase III QAPP (Tierra 2016).

The integrated results of the SQT and Phase III surface sediment sampling program are presented in Table 4-4. An evaluation of the effects of organic carbon normalization on organic COPEC concentrations is presented in Table 4-5.

In general, the collective SQT and Phase III surface sediment data indicate that metals concentrations tend to be highest in the southern portion of the Bay and industrial waterfront areas, while concentrations of organic compounds are more variable between geographic/geomorphic areas. The PCDDs/PCDFs and PAHs tend to be higher in the northern portion of the Bay, while PCB and DDT concentrations are higher in the southern portion of the Bay. A shown in Table 4-5, organic carbon normalization has little effect on explaining the variances in concentrations of hydrophobic organic COPECs (e.g., PCDDs/PCDFs, PCBs, pesticides, etc.) in surface sediments. The coefficients of variation between non-normalized and normalized concentrations would be expected to vary substantially if normalization was driving the spatial difference in COPEC concentrations. Given that this is not the case, no additional analyses on such normalization is utilized in the risk analyses of this BERA. Similarly, as discussed in the RI report, geochemical normalizations explain little of the variability in their concentrations in NBSA sediments.

4.3.5 Sediment Toxicity Data

Sediment samples collected from the 30 SQT locations were subject to two toxicity tests using the amphipod *Leptocheirus plumulosus*. These included a short-term 10-day exposure for survival and a longer-term 28-day exposure for survival, growth, and reproduction (GSH 2017c). The results of the toxicity program are presented and evaluated in detail in the SQT assessment (Appendix A) and Section 6.

4.3.6 Porewater Chemistry Data

An ex-situ laboratory porewater passive sampling program was conducted on field-collected sediments by researchers at the University of Maryland Baltimore County, in coordination with Arcadis and Tierra. Passive samplers for select organic compounds included polyoxymethylene for PCDDs/PCDFs and polyethylene for PCB congeners, pesticides, and semivolatile organic compounds via select ion monitoring. Strips of these samplers were deployed in sediments to extract compounds from the porewater fraction of the sediment. The organic compounds were then extracted from the samplers and the resulting data were used to estimate concentrations of these compounds in the porewater. The estimation procedures are summarized in Appendix B of the Sediment Quality Triad and Porewater Data

Report (GSH 2017d). The calculations rely on the use of a poly-water partition constant (K_{PS}), which is estimated from other studies. Therefore, organic compound porewater concentrations are subject to uncertainty. Large-volume (0.5 liter) diffusion bags were used to capture porewater for direct analysis of metals and other inorganics (i.e., mercury, methylmercury, dissolved organic carbon, ammonia, and total sulfide). A total of 30 porewater samples were analyzed using these methods.

The results of the porewater sampling program are presented in Table 4-6. In general, a majority of inorganic/metal COPECs (arsenic, cadmium, copper, lead, nickel, selenium, silver, and zinc) were characterized by either low detection frequencies or no detection at all in porewater. Pesticides, alpha-BHC and beta-BHC, also exhibited low and no detection respectively. A detailed porewater data evaluation with respect to concentrations and risks to benthic communities is presented in the SQT assessment (Appendix A; Section 6). A discussion of the uncertainty in porewater values is found in Section 6.4.1.

4.3.7 Tissue Chemistry Data

Biota sampling was conducted as part of sampling activities for the BHHRA and BERA to support the risk assessments, and as input parameters to the food web model in this BERA. Given the general open water habitat of the system, and the fact that the biota in the Bay are primarily migratory in their life histories, discerning statistical trends on a broad geographic scale is most relevant. In addition, while it is likely that migratory fish have some background level of contamination in their tissues related to integrated exposures that occur while they are outside Newark Bay, the assumption for this characterization is that their residence time in the NBSA on an annual basis is sufficient for these organisms to achieve an equilibrium with respect to uptake of the primary COPCs in the system. This assumption is evaluated further in the uncertainties sections of this BERA.

4.3.7.1 Polychaete Bioaccumulation Tests

Bioaccumulation tests (laboratory exposures of polychaete worms using field-collected sediments) were conducted at a subset of eight locations in the sub-tidal and intertidal mudflats (Figure 4-2). Polychaete tissue samples were analyzed for the parameter groups listed in Table 4-2. The results of the polychaete tissue analyses are presented in Table 4-7a.

4.3.7.2 Crab and Clam Sampling Program

As part of the BHHRA and BERA investigations, blue crab, softshell clam, and co-located surface sediment sampling were conducted in late summer/early fall 2014. This work was performed in accordance with the Crab and Clam QAPP (Tierra 2014a).

Crab tissue samples collected as part of this program included hepatopancreas, edible muscle, and carcass samples. A total of 37 samples of each crab tissue type were collected, and 18 clam samples were collected. Tissue samples were analyzed for the parameter groups listed in Table 4-2. Additionally, 18 co-located surface sediment samples were collected from the softshell clam locations in intertidal areas by compositing the sediment associated with each clam burrow from which clams were collected at a location. Sediment samples were analyzed for the parameter groups listed in Table 4-2. Crab and clam and surface sediment sample locations are shown on Figure 2-5.

Details of the field program are provided in the Newark Bay Study Area Environmental Sample Collection Report (Normandeau 2017). Results for the crab and clam sampling program are provided in the Crab and Clam Data Report (GSH 2017b). Field, laboratory, data QA/QC procedures, and data validation were also implemented, as specified in the Crab and Clam QAPP (Tierra 2014a).

The results of the clam and crab tissue analyses are presented in Tables 4-7b and 4-7c, respectively. The results of the co-located surface sediment analyses are presented as part of the Phase III sediment data results in Table 4-4. A focused discussion on the concentrations of COPECs in clam and crab and their relative concentrations in the BERA exposure areas is provided in Section 6.

4.3.7.3 Fish Sampling Program

A fish sampling program was conducted as part of the BHHRA and BERA activities, which included a community survey, fish tissue sampling, and a fish health study. These activities were conducted in accordance with the Fish Sampling and Analysis QAPP (Tierra 2014b). A total of 44 fish species were collected over sampling events conducted in three phases (fall 2014, spring/summer 2015, and spring 2016) (Normandeau 2017). Details on the breakdown of species captured by event are provided in Section 2.4.2.

Fish tissue samples were analyzed for the parameter groups shown in Table 4-2. Information regarding fish species collected and analytical results are reported in the Fish Data Report (Tierra 2017). Fish were collected from various locations along the subtidal flats, transitional slopes, and navigational channel throughout each of the three zones (north, south, central) to provide spatial coverage across the NBSA. Sample locations for 2014, 2015, and 2016 are shown on Figures 2-8, 2-9, and 2-10, respectively. The results of the fish tissue analyses are presented in Tables 4-7d-k. A focused discussion on the concentrations of COPECs in various fish species and their relative concentrations in the BERA exposure areas is provided in Section 7.

An evaluation of the effects of lipid normalization on organic COPEC concentrations in biota tissue is presented in Table 4-8. As shown in Table 4-8, lipid normalization has little effect on explaining the variances in concentrations of hydrophobic organic COPECs (e.g., PCDDs/PCDFs, PCBs, pesticides, etc.) in the various species or biota tissue types. The coefficients of variation between non-normalized and normalized concentrations would be expected to vary substantially if normalization was driving the spatial difference in COPEC concentrations. Given that this is not the case, no additional analyses on lipid normalization is utilized in the risk analyses of this BERA.

Additional data were also collected to provide qualitative information about the health of fish in the NBSA. During fish tissue sampling, non-forage fish were externally examined, and any gross physical abnormalities, lesions, or anomalies were documented. A subset of individual forage fish collected were externally examined. Additionally, a representative sample of individual non-forage fish were subjected to internal pathological examination in 2014 and 2015. The fish pathology evaluations included overall morphology, gonad condition, presence of lesions, gonadosomatic index, and internal physical conditions/abnormalities.

Information regarding the fish health results, as well as details of the analytical program, are reported in the Fish Data Report (Tierra 2017). Field, laboratory, data QA/QC procedures, and data validation were also implemented, as specified in the Fish Sampling and Analysis QAPP (Tierra 2014b).

4.3.8 Biological Survey Data

Biological survey data were collected in a variety of ways under the Phase III sampling program. Benthic invertebrate community surveys were conducted as part of the SQT and porewater program. The results of these surveys were presented and discussed in context of historical community data in Section 2.3. A detailed evaluation of the benthic community data with respect to potential risks from COPECs is presented in the SQT assessment (Appendix A) and Section 6.

Semi-quantitative fish and shellfish community surveys were conducted as part of the tissue collection program conducted between 2014 and 2016. The results of these surveys were presented and discussed in context of historical community data in Section 2.4. The details of the field sampling program for benthic invertebrates and fish are provided in Normandeau (2017) (Attachment 4).

4.4 Data Summary for COPECs and BERA Assessment Zones

In this section, the surface sediment and biota tissue data for the COPECs identified in Section 3 are presented for the risk assessment zones identified in Section 3.4. The UCLs of these data, which represent the input parameters for the food web modeling and risk calculations (as discussed in further in Sections 6 through 9), are also presented.

The Bay-wide versus zone-specific data are discussed for sediment, invertebrate, and forage fish; and Bay-wide versus geographic data are discussed for upper trophic level fish species. Sediment data are also discussed in comparison to urban background data compiled from Jamaica Bay under USEPA's REMAP.

Additional data groupings for estimating exposure point concentrations for the risk calculations are provided in respective sections of this BERA as needed.

4.4.1 Surface Sediment, Invertebrates & Fish

The number of sediment and biota samples collected in each BERA assessment zone are listed in Table 4-9. Table 4-10 and Figure 4-4 present the samples locations in each assessment zone for sediment, as well as the sample locations that are considered intertidal mudflat habitat or shoreline. Table 4-11 presents the sample locations in each assessment zone for biota. Zone-specific sediment data summaries for the COPECs are presented in Table 4-12. Boxplots by COPEC that illustrate the distribution of the data for sediments and lower trophic level invertebrate and fish receptors (i.e., clam, blue crab, and mummichog/killifish) are presented in Figure 4-5, and those for upper trophic level fish in Figure 4-6.

In general, there are apparent zone-specific differences in the mean concentrations of many COPECs, including 2,3,7,8-TCDD, total PCBs, metals, and pesticides (total DDx, total Chlordane, dieldrin, hexachlorobenzene), in surface sediments and the tissues of lower trophic level organisms. Average concentrations of most metals, total PCBs, total PAHs, and pesticides are relatively higher in the Southwest Zone than they are in either the North or Southeast Zones. Mean concentrations of 2,3,7,8-TCDD and related dioxin/furan TEQs are generally higher in the North Zone.

The ten composite sediment samples collected in the regularly dredged deep water channels are included in the NBSA-wide dataset, but are not included in the BERA assessment zone datasets and the channels were not evaluated as a separate assessment zone for risk assessment purposes due to the changing conditions in these areas from regular removal of sediments during navigation dredging activities. Summary statistics for these samples are presented in Table 4-12. Mean and UCL concentrations for most of the COPECs including 2,3,7,8-TCDD, total PCBs, PAHs, some pesticides, and most metals are generally lower in the channel sediments than those in the other zones. Exceptions include total DDx, chlordane, mercury/methylmercury, silver, and zinc for which the average and UCL concentrations are within the range of the other zones.

Table 4-9. Sediment and Biota Sample Numbers by BERA Assessment Zone

Matrix/Species	Number of Samples by Assessment ~~Zone[1]~~Zone[a]											
	NBSA-~~wide[5]~~ wide[b]	NBSA-wide Mudflat Only	NBSA-wide Shoreline Only	North	South-east	South-west	North Mudflat Only	South-east Mudflat Only	South-west Mudflat Only	North Shoreline Only	South-east Shoreline Only	South-west Shoreline Only
Sediment	254	21	51	104	50	90	13	3	5	21	14	16
Polychaete	8	--	--	~~--~~4	~~--~~2	~~--~~2	--	--	--	--	--	--
Softshell Clam	18	--	--	8	4	6	--	--	--	--	--	--
Blue Crab	37	--	--	12	13	12	--	--	--	--	--	--
Mummichog/ Killifish	17	--	--	8	3	6	--	--	--	--	--	--
White Perch	27	--	--	9	9	9	--	--	--	--	--	--
Summer Flounder	8	--	--	2	2	4	--	--	--	--	--	--
Winter Flounder	10	--	--	4	4	2	--	--	--	--	--	--
American Eel	18	--	--	6	6	6	--	--	--	--	--	--
Atlantic Menhaden	2	--	--	2	--	--	--	--	--	--	--	--
Atlantic Silverside	7	--	--	1	3	3	--	--	--	--	--	--

Notes: ~~[1]Sample~~
[a]Sample IDs and Assessment Zone assignments can be found in Tables 4-10 and 4-11.
[b]The NBSA-wide samples include the ten composite samples collected in the regularly dredged deep water channels. The 10 composite samples were not included in the BERA assessment zones (i.e., North, Southeast, Southwest).

ANOVA models were used to evaluate the differences in mean concentrations in sediment (Table 4-13a) and whole body blue crab (Table 4-13b) among the BERA assessment zones (North, Southeast, Southwest). Sediment concentrations of hexachlorobenzene, total DDx and most metals are significantly greater in the Southwest Zone (Table 4-13a) while 2,3,7,8-TCDD is significantly greater in the North Zone. For whole body blue crab, concentrations of total DDx and arsenic were significantly greater in in the Southwest Zone (Table 4-13b) while 2,3,7,8-TCDD, PCBs, pesticides other than DDx, PAHs, mercury and selenium were significantly greater in the North Zone. The risk implications for the zone-specific differences are examined by receptor in Sections 6 through 9.

~~[5] The NBSA-wide samples include the ten composite samples collected in the regularly dredged deep water channels. The 10 composite samples were not included in the BERA assessment zones.~~

4.4.2 Evaluation of Urban Background

A comparison of the NBSA surface sediment data from the Phase III RI/BERA sampling programs to data collected in Jamaica Bay under the REMAP program in 2008 and 2013 is presented in Table 4-14. This analysis was conducted to evaluate whether any COPECs identified for the NBSA may be considered comparable to urban background levels for the NY/NJ Harbor Estuary. Jamaica Bay is considered by USEPA to be the least impacted area in the Harbor Estuary, with very limited potential influence from legacy CERCLA sites within the embayment. This area was used as the marine/estuarine urban background location for the LPRSA RI as well.

In general, the data indicate that most of the COPECs in the NBSA are elevated relative to Jamaica Bay. The exceptions are selenium and silver, which have similar mean concentrations in the NBSA sediments to those in Jamaica Bay. The mean metals concentrations were 1.1 (silver) to 7.2 (mercury) times greater in the NBSA than in Jamaica Bay, with many exceedances of the upper 95th percentile of the Jamaica Bay concentrations noted (Table 4-14). 2,3,7,8-TCDD concentrations are substantially elevated in the NBSA compared to Jamaica Bay (mean ratio of 85). Total Aroclor PCBs were infrequently detected in Jamaica Bay; however, 90% of the NBSA concentrations exceeded the Jamaica Bay maximum. Pesticides were also infrequently detected in Jamaica Bay and ~~detection~~quantitation limits tended to be greater than those in the Phase III data, making comparison difficult. However, the statistics suggest that hexachlorobenzene and total DDx are elevated in the NBSA compared to Jamaica Bay. Total PAHs (total HMW, total LMW, and total) are elevated in the NBSA compared to Jamaica Bay, with ratios of means ranging from 26% to 32% and 85% to 88% of Phase III concentrations exceeding the 95th percentile of the Jamaica Bay concentrations (Table 4-14).

The results of this analysis help to provide some perspective on the COPECs in Newark Bay in terms of distinguishing those that are comparable to urban background for the NY/NJ Harbor Estuary (e.g., selenium and silver), versus those that are clearly elevated in the NBSA. While somewhat limited in its overall usefulness due to analytical limitations of the REMAP Jamaica Bay dataset identified above for many COPECs, this information can be used as a line of evidence to inform decision-making regarding risk management and remediation options. There are no comparable invertebrate or fish tissue background datasets available. Some limited perspectives are provided on fish tissue background where applicable in Section 6.

4.5 Whole-Body Crab Calculation

Blue crab tissue was analysed as three separate tissue types: edible muscle, hepatopancreas and carcass. Tissue types were precisely weighed ~~for thirty-four~~in 34 individual crabs collected September 10-11, 2014 (Normandeau 2017). After that date, for efficiency, the field processing protocol was changed such that the individual tissue types for each individual crab were no longer weighed separately. An average of the fractional weights (Table 4-15) for these 34 individually and precisely measured crabs was calculated ~~in order~~and used to combine the tissue types for estimation of the whole-body crab concentration for all sample composites collected in the NBSA. The whole-body crab concentration was calculated as follows:

$$C_{WB} = (C_{mus} \times f_{mus}) + (C_{hep} \times f_{hep}) + (C_{car} \times f_{car})$$

where,

C_{WB} = estimated whole-body crab concentration

C_{mus} = muscle concentration

f_{mus} = fraction of whole-body weight that is muscle (i.e., 0.54)

C_{hep} = hepatopancreas concentration

f_{hep} = fraction of whole-body weight that is hepatopancreas (i.e., 0.19)

C_{car} = carcass concentration

f_{car} = fraction of whole-body weight that is carcass (i.e., 0.27)

When one or two of the three tissue types within a sample had non-detect concentrations, one-half the ~~reporting~~quantitation limit was substituted as the concentration for the non-detect tissue type. When all three tissue types were non-detect, the whole-body concentration was reported as a non-detect at the lowest ~~reporting~~quantitation limit of the three tissue types.

4.6 Historical Data

Historical studies have been conducted to understand the contaminant distribution in water, sediment, and biota in the Bay and surrounding areas, and their potential effects on humans and the Bay ecosystem. Throughout the RI process, searches have been performed to identify, locate, and evaluate such studies and determine their usefulness to support the RI and risk assessments. Details and data summaries for these secondary datasets are provided in the Phase I and Phase II Data Evaluation and Analysis Report (Tierra 2014c). Table 4-16 provides an overview of the available secondary data.

The uses of these secondary datasets in this BERA are generally limited to characterizing the ecology of the NBSA and providing historical perspective on potential changes in conditions and related risks in the system. However, several toxicological studies that have been conducted on invertebrates, fish and birds in Newark Bay are used in this risk assessment to inform the development of TRVs for risk assessment, and to provide perspectives on potential COPEC-related effects that have been observed on these organisms. Concentrations in fish and crabs for selected COPECs measured in 1999 and 2000 in the NBSA as part of the Contamination Assessment and Reduction Program by NYSDEC are shown in Table 4-17.

5 SURFACE WATER ASSESSMENT

Surface water in the NBSA was monitored between 2010 and 2013, with more than 200 samples collected and analyzed for typical water quality parameters, as well as the chemical constituents listed in Table 4-2. Summary statistics for the COPEC analytical results are presented in Table 5-1. These results are compared to their respective New Jersey Surface Water Quality Criteria (SWQC) for aquatic life, as well as various conservative ecological screening values from the scientific literature. The state use designation of Newark Bay is Saline Estuarine Category 3 (SE3) waters. The primary uses of SE3 waters are for secondary contact recreation (i.e., boating, fishing), and primarily maintenance of the waters for migratory fish and wildlife use and propagation. The aquatic life criteria are a conservative means to evaluate the water quality of the Bay for ecological uses. The goal of this evaluation is to evaluate whether surface water likely represents a substantive COPEC exposure pathway or potential risk to ecological receptors.

Concentrations of metals and organic COPECs, with the exception of copper in a few samples (8 out of 274 samples) were below their respective New Jersey SWQC (Table 5-1). Boxplots (Figure 5-1) illustrate the distributions of COPC concentrations, which are characterized by tight ranges with a few outliers. Overall, the low concentrations of COPCs in surface water do not exhibit much variability and are consistent with respect to space, time, flow regime, depth, or tidal cycle. Because contaminants tend to preferentially associate with solids, the low variability in COPC concentration is indicative of low variability in suspended solids concentrations in the Bay. A more detailed discussion of the surface water data is presented in the RI Report (GSH ~~2019, in preparation~~2020).

Given the fact that COPEC concentrations in surface water do not exceed either NJ SWQC or other conservative surface water screening guidelines/benchmarks, and the concentrations of COPECs are similar spatially throughout the Bay (vertically, tidally and seasonally) it is highly unlikely that:

- COPECs in surface water pose a risk to aquatic communities, including plankton, algae, shellfish, or fish
- Surface water represents a substantive exposure pathway for uptake of bioaccumulative COPECs (relative to those from sediments and food web exposures) for wildlife.

The conservative nature of this water screen is comparable to that of the SLERA, and therefore the conclusion of no risk has a high degree of certainty. For these reasons, surface water is not further evaluated in the SQT assessment for benthic invertebrates, or in subsequent exposure modeling of COPECs for semi-aquatic wildlife receptors. Instead, the focus of this BERA is on exposures from sediment- and food web-associated COPECs.

Uncertainty in the evaluation pertains to the toxicity of DDx isomers. While no exceedances of the chronic SWQC for 4,4'-DDT (i.e., 1 nanogram per liter [ng/L]) were observed, either for 4,4'-DDT itself or for the other 4,4' isomers individually, exceedances do occur when the sum of DDx isomers is compared to the chronic 4,4'-DDT SWQC (Table 5-1). There are 38 exceedances (24% of all observations) for total 4,4' ~~DDx; however, the overall mean (i.e., 0.85 ng/L) is less than the chronic 4,4'-DDT SWQC.~~ -DDx; however, the overall NBSA-wide mean (i.e., 0.85 ng/L) is less than the chronic 4,4'-DDT SWQC. There are some spatial differences among the NBSA stations. At station ARK, located at the mouth of the Arthur Kill

(Figure 4-1), mean total 4,4'-DDx was 1.6 ng/L which exceeds the chronic SWQC for 4,4'-DDT. In addition, 50% of the 28 total 4,4'-DDx results from this station exceeded the chronic 4,4'-DDT SWQC. The highest DDx concentrations were noted during the December 2012 sampling event, which took place during average tidal and normal flow conditions (i.e., as measured at Dundee Dam in the Passaic River). At all other NBSA stations, mean total 4,4'-DDx concentrations were less than 1 ng/L.

Surface water TRVs for the evaluation of risk to fish were derived for the LPRSA BERA (Windward 2019). An estuarine chronic TRV of 300 ng/L for 4,4'-DDE was based on the lowest chronic toxicity value available in the ECOTOX database (USEPA 2016b), which was for a copepod (*N. spinipes*). Surface water concentrations of 4,4'-DDE in the NBSA are orders of magnitude lower than this TRV (Table 5-1). An estuarine chronic TRV was also derived for 4,4'-DDT/total DDx based on a species sensitivity distribution that included data for 14 fish and 18 invertebrate species (Windward 2019). This value, 19 ng/L, exceeds all observed surface water concentrations of total 4,4'-DDx in the NBSA (Table 5-1).

6 BENTHIC INVERTEBRATE ASSESSMENT

Benthic invertebrates in the NBSA include both infaunal communities and several epibenthic shellfish such as softshell clam, Baltic clam, eastern oyster, and blue crab. There are a variety of taxonomic feeding guilds for benthic and epibenthic invertebrates including filter-feeders, deposit-feeders, detritivores, and omnivores. These organisms live in close association with the sediments in the Bay and, therefore, represent some of the highest potential for exposure to COPECs in sediments. Invertebrates also form the base of the food web for fish and other upper-trophic level organisms such a birds and mammals. This section includes the risk assessment for invertebrates in the NBSA.

The following are the Assessment Endpoints and Risk Questions for invertebrates that are addressed in this section.

Assessment Endpoint: Survival, growth, and/or reproduction of invertebrates.

Risk Question 1: *Are invertebrate communities in the NBSA different from those found in similar nearby water bodies with chemical concentrations at regional background levels?*

Risk Question 2: *Are the COPEC concentrations in sediments from the BAZ greater than benchmarks for the survival, growth, and/or reproduction of invertebrates?*

Risk Question 3: *Is the survival, growth, and/or reproduction of invertebrates exposed to sediments from the BAZ of the NBSA significantly lower than that in reference sediments?*

Risk Question 4: *Are COPEC concentrations in porewater from the NBSA greater than toxicological benchmarks for the survival, growth, and/or reproduction of invertebrates?*

Risk Question 5: *Are COPEC residues in invertebrate tissues from the NBSA greater than tissue (whole-body) toxicity benchmarks for the survival, growth, and/or reproduction of invertebrates?*

Risk Questions 1 through 4 are addressed by the SQT Assessment presented in Section 6.1. Risk Question 5 is addressed in the tissue-based assessment in Section 6.2. A detailed description of the benthic communities that inhabit the NBSA (including perspectives on historical changes in the communities) is provided in Section 2.3. The life history profiles for softshell clam and blue crab are provided in Appendix B.

6.1 Sediment Quality Triad Assessment

SQT data were collected from 30 locations in the NBSA (Figure 6-1). Sediment grab samples from each of these locations were prepared for the following analyses: sediment chemistry, toxicity testing (10 and 28-day *L. plumulosus* whole sediment bioassays), benthic invertebrate taxa enumeration, and porewater chemistry. Sediment and porewater samples were analyzed for the parameter groups listed in Table 4-2. These data were collected for the purpose of evaluating whether exposure to site-related COPECs in the NBSA poses unacceptable risks to benthic invertebrate communities (BIC).

The SQT program was designed to support an effects-based, WOE assessment to evaluate risks to benthic invertebrate communities using three primary LOEs: 1) sediment chemistry, 2) sediment toxicity, and 3) benthic invertebrate community metrics. The approach is widely utilized by regulatory agencies,

including USEPA Region 2 and its Partner Agencies, as the most appropriate process currently available for assessing potential risks to benthic invertebrates. It is the approach that is being used to assess risks to benthic invertebrates for the LPRSA. For consistency purposes and data comparability, this approach is also used for the NBSA.

The SQT method incorporates a combination of both quantitative and qualitative analyses to identify potential cause-and-effect relationships between two or more of the components (Tierra 2015b). The specific technical approach for the SQT was developed by GSH and USEPA in an iterative and collaborative manner between 2016 and 2018.

The SQT evaluation methods include comparisons of BIC and toxicity data to an appropriate reference site. The reference site chosen for this evaluation is the Jamaica Bay Estuary, New York, which is physically similar to the open water habitats of the NBSA. A total of 150 SQT samples (samples with co-located chemistry, toxicity [i.e., 10-day *A. abdita* bioassay], and BIC data) have been collected from Jamaica Bay over of period of years (1993 through 2013) by USEPA under REMAP and by other agencies. A screening process was conducted to select a subset of these 150 samples to represent reference conditions for the benthic invertebrate community (Appendix A). The Jamaica Bay sediment chemistry data were compared to effects range-low (ERL) and effects range-median (ERM) values (Long et al. 1995). Acceptable Jamaica Bay reference locations had three or fewer exceedances of ERLs and no exceedances of ERMs across all chemicals for which those sediment guidelines were available. In addition to meeting the chemical criteria, acceptable reference locations in Jamaica Bay were required to meet sediment toxicity criteria. Specifically, *A. abdita* survival results at Jamaica Bay reference locations were required to be ≥ 80% of the respective negative control response. The screening evaluation resulted in a total of 59 SQT samples from Jamaica Bay suitable for use as reference data in a reference envelope evaluation. The Jamaica Bay reference data and screening results are provided in Appendix A.

The SQT analysis methods fall into two major categories:

- The SQT evaluation (Section 6.1.1) is an objective method of comparing sediment chemistry concentrations, sediment toxicity endpoints, and BIC metrics to screening, control, or reference values with the objective of categorizing each sampling station with respect to the degree of potential impact.
- The quantitative effects assessment (Section 6.1.2) incorporates bivariate and multivariate statistical methods to evaluate the association of sediment and porewater chemistry with effects measures (i.e., toxicity endpoints and BIC metrics).

Appendix A provides the detailed results and calculations of the comprehensive analyses of the SQT data described here.

6.1.1 SQT Evaluation

This evaluation was used to classify each of the 30 SQT stations with respect to degree of potential impact. Three categories (i.e., the triad) were evaluated: BIC composition, sediment toxicity, and sediment/porewater chemistry. Each category has equal weighting in the evaluation and was assigned a score ranging from 0 (unimpacted) to 1 (most impacted). The scores for each category are summed (maximum possible score = 3) and the final score is used to categorize each station with respect to

degree of impact (Table 6-1). This methodology is consistent with the SQT evaluation of the LPRSA (Windward 2019).

The following three subsections discuss how the evaluation and scoring was conducted within each of the triad categories. Section 6.1.1.4 discusses the final classification of the SQT stations.

6.1.1.1 BIC

Benthic invertebrate communities were characterized by identifying and enumerating the benthic invertebrates from sediment grab samples (three replicates per station) collected at the 30 SQT stations. Taxa in BIC samples were identified to the lowest practicable identification level, in a manner consistent with other surveys performed in the NY/NJ Harbor Estuary (Weisberg et al. 1998). The BIC taxonomic data are presented in Appendix A. The following five standardized BIC metrics were calculated for each sample as described in Appendix A:

- Abundance (i.e., density): the number of organisms per square meter
- Number of Taxa (i.e., number of discrete lowest practicable identification level taxa)
- Shannon-Weiner diversity index (Shannon 1948)
- Pielou's evenness index (Pielou 1966)
- Swartz's dominance index (Swartz et al. 1985).

In addition, a multimetric B-IBI was calculated using a method derived for the saline portions of the NY/NJ Harbor Estuary (Weisberg et al. 1998). Threshold values of the B-IBI for the classification of impacts are as follows: 1 to <2 = impacted, 2 to <3 = slightly impacted, and 3 to 5 = unimpacted (Weisberg et al. 1998). These six BIC metrics were also used in the quantitative effects assessment described in Section 6.1.2.

The BIC data from each SQT station were evaluated and scored using a reference envelope approach (Table 6-1). This approach involves the comparison of metrics measured at the potentially impacted SQT stations to lower (or upper) percentiles of the distribution of metric values measured at the reference site. Four of the five standardized BIC metrics (number of taxa, Shannon-Weiner diversity index, Pielou's evenness index, and Swartz's dominance index) are expected to decline in value as a result of physical and/or chemical degradation. Therefore, these metrics were compared to the 5th percentile of the Jamaica Bay reference distribution. BIC metrics that are equal to or greater than the 5th percentile of the reference distribution are considered comparable to reference. Abundance is a metric that can either decline or increase as a result of degradation (Weisberg et al. 1998). Therefore, abundance was compared to both the 5th and the 95th percentile of the Jamaica Bay distribution. Abundance equal to or greater than the 5th percentile and less than or equal to the 95th percentile of the Jamaica Bay reference distribution is considered comparable to reference. Metrics outside the reference envelope received a score of 1 (Table 6-1). For the B-IBI, a score of either 0, 0.5, or 1 was applied according to the impact categories described by the authors (Weisberg et al.1998) with respect to degree of impact (Table 6-1). Each metric score was weighted equally (0.167) and the weights were multiplied by the metric score and summed, such that the maximum possible score for the BIC is 1, representing the greatest impact.

Table 6-2 presents the metric values and SQT scores for each of the 30 stations. The Jamaica Bay reference envelope values are shown in the top row for each metric. (The details of the percentile calculations are provided in Appendix A.) For the five standardized metrics only one value, abundance at station 148, was outside the Jamaica Bay reference envelope. Based on the B-IBI values (Table 2-9), 21 stations were classified as non-impacted (Table 6-2) and eight stations were classified as slightly impacted (three in the north, one in the central, and four in the south). Station 148, which is located along a bulkheaded portion of the southern shoreline of the NBSA (Figure 4-2), was classified as impacted. The B-IBI scores are depicted on Figure 2-4. The combined BIC SQT scores are very low for most stations indicating non-impacted BIC. The BIC at station 148 exhibited low abundance and number of taxa along with a low B-IBI score indicating potential impact to BIC at that location.

To further support the comparison of BIC metrics to the Jamaica Bay reference data, hypothesis testing was also conducted (Table 6-3). Mean abundance in Jamaica Bay is greater than that of the NBSA. Because abundance can decrease or increase with degradation, inferences regarding comparability are uncertain for this metric and no hypothesis testing was conducted. For the other metrics, the Jamaica Bay reference means were compared to the NBSA means using the non-parametric Kruskal-Wallis test. The mean number of taxa in Jamaica Bay (19) is lower than that of the NBSA (21-26), but the difference is not significant. For other metrics, Shannon Wiener diversity, Pielou's evenness, and Swartz's dominance, the Jamaica Bay mean is significantly lower that the NBSA mean (Table 6-3). These results support the reference envelope comparison findings and indicate that the BIC in the NBSA are comparable to the BIC in the Jamaica Bay reference. Boxplots of the BIC metric data can be found in Appendix A (Figures A-3 and A-4).

6.1.1.2 Toxicity

Ten-day acute and 28-day chronic toxicity tests were conducted on sediment from each of the 30 SQT stations using the amphipod *L. plumulosus* (USEPA 2001).

The following endpoints were measured for each sample:

- 10-day survival
- 28-day survival
- Growth rate (milligrams per organism per day) at Day 28
- Reproduction (young per surviving adult) at Day 28.

Five replicates with 20 amphipods each were tested for each sample and a laboratory control sample. For each endpoint, the mean response for each sample was compared to the laboratory control response to determine if the sample response was statistically significantly different than the control response (α = 0.05) using the methods described in the test guidance (USEPA 2001). Ten- and 28-day survival results and statistical significance are depicted in Figure 6-2.

The sediment toxicity data comprise survival and sublethal (i.e., growth and reproduction) endpoints. Criteria for assigning toxicity are well established for the survival endpoints, but not for the sublethal endpoints. Criteria used for REMAP (USEPA 2003a) defined samples as toxic if the 10-day amphipod (i.e., *A. abdita*) survival was less than 80% of the control survival and highly toxic if it was less than 60% of the control survival. The 80% decision criterion is consistent with the criterion in the testing manual for

discharge to U.S. waters (USEPA and USACE 1998) and was also suggested by Kennedy et al. (2009) for 28-day *L. plumulosus* toxicity tests. Therefore, these criteria were adopted to define toxicity of sediments from the NBSA based on the survival endpoints. Where survival is ≥ 80% of control and/or not significantly different from control, a score of 0 was applied (Table 6-1). Survival significantly different from control and between 60 and 80% of control received a score of 0.5. Survival significantly different from control and < 60% of control received a score of 1 (Table 6-1).

The sublethal endpoints are subject to considerable response variability compared to the survival endpoints. Eickhoff et al. (2014) subjected five control sediments to 28-day tests with *L. plumulosus* and found little variation in survival among the five control samples, which ranged from 96 to 99%. Within-batch replicate variance was also low for survival, with a mean coefficient of variation (CV) of 5.6%. However, growth and reproduction endpoints were more variable both among control samples and within batch replicates. Mean growth in control samples ranged from 0.7 to 1.8 milligrams per organism and the mean CV among replicates was 30%. Reproduction had a 10-fold range within control sediments (0.8 to 8.4 juveniles per surviving adult) and the mean CV among replicates was 79%. Kennedy et al. (2009) found significant variability in control sediment for growth (CV = 56%) and reproduction (CV = 63%), as well as significant variability in an interlaboratory comparison of the growth and reproduction endpoints. *L. plumulosus* reproduction is known to be highly dependent on grain size (McDonald et al. 2010). This fact was illustrated in Kennedy et al. (2009) where it was found that a clean reference sample, which was relatively coarse-grained, had a very low response for the reproductive endpoint compared to the control (i.e., less than 10% of the control response). Based on the typical criteria used to define toxicity for survival endpoints (i.e., less than 80% of the control response), the clean reference sample would be classified as toxic based on the reproductive endpoint.

Given the large variability that can occur among control sediments for these sublethal endpoints and because there are no established control acceptability criteria for these sublethal endpoints, a simple comparison to control sediment cannot be used to judge a sediment sample as toxic. Further, high variability makes default application of decision criteria (e.g., 20% reduction compared to control) inappropriate (McDonald et al. 2010). Given these considerations, the SQT scoring for the sublethal endpoints was adjusted as compared to the survival endpoints. Where sublethal response is ≥ 70% of control and/or not significantly different from control, a score of 0 was applied (Table 6-1). Sublethal response significantly different from control and between 50 and 70% of control received a score of 0.5. Sublethal response significantly different from control and < 50% of control received a score of 1 (Table 6-1).

The NBSA toxicity data for sublethal endpoints was also subject to high variability (Attachment A-2 of Appendix A). The CV for 28-day survival ranged from 4% to as high as 139% with variance generally increasing as mean survival decreases. ~~This variability can make statistical comparisons with control data challenging.~~ The average minimum detectable difference (MDD) at α = 0.05 and β = 0.20 for this endpoint is ~~27~~33%. This is why some stations with survival less than 80% of control were not found to be statistically significantly different than control. The uncertainty of this with respect to SQT scoring is discussed in Section 6.4.

Each toxicity endpoint score was weighted equally (0.25) and the weights were multiplied by the score and summed, such that the maximum possible score for the toxicity category of the triad is 1, representing the greatest impact. The scoring and weighting are consistent with the SQT evaluation

presented in the BERA conducted for the lower 17 miles of the LPRSA (Windward 2019); however, in that BERA the responses were compared to a reference threshold response whereas for the NBSA BERA, the response is compared to the laboratory control response.

Table 6-4 presents the measured values and scoring for the four sediment toxicity endpoints at each of the 30 SQT stations. Statistical significance is depicted by shading. Values less than the control value minus the minimum detectable difference are shown in bold. Twenty-eight stations had 10-day survival comparable to the laboratory control survival (90%). Only two stations, 151 and 154, located near the southwestern shoreline of the NBSA (Figure 6-2) exhibited acute toxicity and received the maximum SQT score of 1. Thirteen stations received a score of 0 because all endpoints were comparable to control based on the scoring criteria. Fifteen stations had scores between 0 and 1 based on a comparison of chronic endpoints to laboratory control. However, in total, only five stations had 28-day survival less than 80% of control and significantly different than control (151, 154, 143, 158, and 160). Twenty-eight day survival at Station 141 (67%), while significantly different than control, was greater than 80% of control (i.e., 83% of control) and therefore received a score of 0 for this endpoint. Alternative scoring methods are evaluated in the uncertainty analysis.

A comparison of mean 10-day amphipod survival to reference survival was also conducted (Table 6-5) using the non-parametric Kruskal-Wallis test. While two different species were tested in the two estuaries, *A. abdita* in Jamaica Bay and *L. plumulosus* in the NBSA, these two species were found to be similar in their ability to identify toxic sediment samples in a round robin inter-laboratory study (Schlekat et al. 1995). Survival was first adjusted to control survival as follows: Control-adjusted survival = Observed survival/Control survival. The mean control adjusted survival in the NBSA was compared to the mean control adjusted survival in Jamaica Bay. The overall NBSA mean and the means for each geographic area were statistically significantly greater than the Jamaica Bay reference mean. It is important to note that the mean 10-day survival in the Jamaica Bay reference dataset is biased high due to the screening process. All samples with observed survival less than 80% of control were screened from the Jamaica Bay dataset regardless of whether the sample had inflated chemistry. Therefore, the actual mean control survival in Jamaica Bay based on all samples or even just those samples that pass a chemistry screen is lower than that based on the 59 samples that passed the chemistry and toxicity screen. Nonetheless, even when compared to this biased high mean control-adjusted reference survival, the NBSA is demonstrated to be, at the very least, comparable to Jamaica Bay. Boxplots and full details of the hypothesis testing are described in Appendix A. Details of the screening of the 2008 and 2013 REMAP data for Jamaica Bay are found in Table A-1-3 of Appendix A, Of the 37 samples that passed the toxicity screen, 17 did not pass the chemical screen.

6.1.1.3 Sediment and Porewater Chemistry

Sediment and porewater chemistry results at each of the 30 SQT stations were compared to screening values and water quality criteria to develop the score for the chemistry category of the triad (Table 6-1).

6.1.1.3.1 Sediment Scoring

Sediment screening values were preferentially based on those published either by USEPA (2005a) or Long et al. (1995). The USEPA (2005a) developed sediment screening values for a number of chemicals based on their predicted toxicity to amphipods. For each chemical T20, T50, and T80 values were

developed that correspond to probabilities of toxicity of 20%, 50%, and 80%, respectively. Long et al. (1995) developed sediment screening values, ERL and ERM, for a number of chemicals that correspond to threshold and probable effect concentrations. Long et al. (2006) also present a method for assessing sediment quality using an ERM quotient, which is the sum of the chemical concentrations in sediment divided by the ERMs for 25 chemicals. An ERM quotient was calculated for each SQT sample. For SQT scoring, a threshold value was developed based on the T20 and/or the ERL. For chemicals that had both a T20 and an ERL, the maximum of the two values was chosen as the threshold value. Similarly, a median value was developed based on the T50 and/or the ERM. For chemicals that had both a T50 and an ERM, the maximum of the two values was chosen as the median value. For chemicals with a FOD greater than 25% in surface sediment for which T20/T50 and/or ERL/ERM values were not available, threshold and median scoring criteria were based on threshold effect level or probable effect level values for marine sediment published in the NOAA Screening Quick Reference Table (SQuiRT) (Buchman 2008). See Appendix A (Table A-8) for a list of the sediment screening values. If a sediment sample had a concentration for any chemical that exceeded the threshold value, it received a score of 0.5 (Table 6-1). If a sediment sample had a concentration for any compound that exceeded the median value and/or its ERM quotient exceeded 1, it received a score of 1 (Table 6-1). Sediment concentration made up 50% by weight of sediment chemistry score (Table 6-1).

Table 6-6 presents the number of exceedances by chemical for each of the screening values at the 30 SQT sediment samples. The threshold values for most chemicals were exceeded at most stations and therefore threshold values provide very little information in the way of discriminating among stations with respect to potential impact. Median values were also frequently exceeded, again providing little value for discriminating among stations.

The sediment data were also evaluated according to USEPA equilibrium sediment partitioning guidance for metal and PAH mixtures (USEPA 2003b, 2005b, 2017a). Under equilibrium partitioning theory, no toxicity is expected in any of the SQT sediment samples due to divalent metals concentrations. However, the potential for toxicity due to PAHs is predicted at 7 stations. Detailed results are provided in Appendix A (Table A-14).

6.1.1.3.2 Porewater Scoring

Porewater screening values used in the SQT scoring were preferentially based on the New Jersey acute and chronic SWQC for aquatic life in saline water (NJDEP 2016). Because a portion of the NBSA is in the state of New York, the NYSDEC ambient water quality standards were reviewed. NYSDEC applies only acute aquatic water quality criteria to saline waters and the New Jersey SWQC were equal or greater to the NYSDEC standards and are therefore protective based on NYSDEC standards. For detected compounds without WQC, acute and chronic screening values were selected from the NOAA SQuiRT tables for marine surface waters. If marine guidelines were not available, freshwater guidelines were selected. See Appendix A (Table A-11) for a list of screening values for porewater. In addition, a sum of toxic units (SumTU) approach was evaluated for mixtures of PAHs and metals in porewater (see Table A-14 in Appendix A) as described in guidance (USEPA 2003b, 2005b, 2017a). For metals, the SumTU is a unitless value equal to the sum of each of the divalent metals (i.e., cadmium, chromium, nickel, lead, and zinc) concentrations divided by its final chronic value. A sample with a SumTU greater than 1 has the potential to exhibit chronic toxicity (USEPA 2005b). Similarly, for PAH mixtures, the SumTU is equal to the sum of the concentrations of 34 PAHs divided by their final chronic values (USEPA 2017a). If a

porewater sample had a concentration for any chemical that exceeded a chronic criteria or screening value, it received a score 0.5. If either the metal or PAH SumTU of the sample exceeded 1, it received a score of 0.5 regardless of whether any individual chronic criteria were exceeded. If a porewater sample had a concentration for any chemical that exceeded an acute criteria or screening value, it received a score of 1. Porewater concentration made up 50% by weight of the sediment chemistry score (Table 6-1).

Table 6-7 presents the number of exceedances by chemical for each of the screening values for the 30 porewater samples. Exceedances of chronic SWQC were observed for PCBs and total DDx, but all other organic compound concentrations were below chronic screening values. Arsenic, copper, and zinc were infrequently detected in porewater, but where they were detected they were often in exceedance of acute or chronic SWQC. Of the five acute exceedances observed, three were for zinc, and one each was for arsenic and copper. Metals SumTU exceeded 1 at six stations. PAH SumTU exceeded 1 at station 154 and was nearly 1 at station 151, the two stations exhibiting acute toxicity.

6.1.1.3.3 Sediment and Porewater Chemistry Summary

Table 6-8 presents the summary of exceedances and scoring for each of the SQT stations. For sediment concentrations, all stations received the maximum score of 1 due to exceedances of the median sediment screening values or ERM quotients exceeding 1. For porewater concentrations, exceedances of chronic criteria were observed at 18 stations, 5 of which also had an exceedance of acute WQC. The metals SumTU exceeded 1 at six stations. The PAH SumTU exceeded 1 at one station, 154, where acute toxicity was also observed. No porewater exceedances were observed at 12 stations, 11 of which received an SQT score of 0 because neither SumTU exceeded 1. The combined sediment and porewater SQT scores ranged from 0.5 to 1.

For some chemicals, strong relationships were apparent between sediment and porewater concentrations (e.g., 2,3,7,8-TCDD, PCBs, and DDx). For others, such as mercury, the relationship is not very strong. For PAHs, there is no apparent relationship. Bivariate plots of porewater vs. sediment concentrations at the SQT stations are given in Attachment A-4 of Appendix A.

6.1.1.4 Summary of the SQT Evaluation

Table 6-9 presents the final scores for each station based on each leg of the triad. Consistent with the LPRSA BERA (Windward 2019) and as agreed upon with USEPA, the final scores were assigned impact categories as follows: no impact = $\leq$0.75; low impact = >0.75 and $\leq$1.5; medium impact = >1.5 and $\leq$ 2.25; and high impact = >2.25. Most SQT stations had scores indicating no impact (10) or low impact (18). Two stations (151 and 154) were classified as medium impact. While the BIC metrics at these stations did not indicate impact, acute toxicity and elevated chemistry was observed. In fact, these stations had the highest concentrations observed in SQT sediment and porewater samples for many chemicals (Table 6-10). No stations were classified as high impact.

6.1.2 Quantitative Effects Assessment

6.1.2.1 Spearman Rank Correlation

Spearman rank correlation, a non-parametric measure, was used to evaluate the strength of the association between chemical concentrations and effects measures (i.e., toxicity endpoints and BIC metrics). The Spearman rank correlation coefficient is computed with ranked data and describes the strength of the monotonic association between two variables. Correlation coefficients can range in value from -1 to 1 and the absolute value describes the strength of the association with 0 indicating no association and 1 or -1 indicating the strongest association. Positive values indicate one variable increases with the other. Negative values indicate one variable decreases with the other. Correlation coefficients, like all statistics, are subject to indicating an association by chance where none exists and the probability of this is a function of sample size. For a sample size of 30, Spearman correlation coefficients with an absolute value greater than 0.3 indicate statistically significant association ($\alpha = 0.05$).

Tables 6-11 and 6-12 give the Spearman rank correlation coefficients for effects measures and selected sediment and porewater chemical concentrations, respectively. The coefficients for all detected chemicals are found in Appendix A (Tables A-19 and A-20). Coefficients less than -0.3 are bolded to indicate that a chemical has a significant negative association with an effect measure. Many chemicals including metals, PCBs, PCDD/PCDFs, pesticides, and PAHs are significantly and negatively associated with toxicity endpoints in both sediment and porewater. While association does not indicate causation, there is evidence that chemical stressors may be responsible for the observed toxicity. The evidence is less clear for BIC metrics. For example, while several sediment chemical concentrations are negatively correlated with number of taxa, this association is not seen for porewater.

Table 6-13 summarizes the chemicals of potential concern to the BIC based on the SQT evaluation and the correlation analysis. Chemicals on this list had exceedances of sediment and/or water quality guidelines/criteria and also were correlated with toxicity and/or BIC metrics for either sediment or porewater.

6.1.2.2 Multivariate Analysis

Because many chemicals are associated with effects measures, dose response modeling of individual chemicals would not be appropriate. Therefore, multivariate analyses were conducted to further evaluate the relationships between chemicals and effects. One method of multivariate analysis is multivariate regression. In this method, multiple independent or predictor variables (e.g., chemical concentrations) can be evaluated with respect to their association with dependent or response variables (e.g., effects measures). However, when the predictor variables in a regression model are highly correlated, parameter estimates are unstable and have high standard errors, a problem referred to as collinearity or multicollinearity (Jackson 1991). A solution to this problem is to transform the predictor variables into principal components and use the principal components in subsequent modeling of the response variables (Jackson 1991). This transformation, called principal component analysis (PCA), was used to evaluate the relationships between sediment and porewater chemistry concentrations and response variables because these chemicals are in fact highly correlated in the NBSA.

In PCA, a multivariate dataset is transformed using matrix algebra (i.e., orthogonal rotation) to create a new set of uncorrelated variables known as principal components (PCs), which are linear combinations of the original variables (i.e., chemical concentrations). PCs are sorted such that each, in turn, represents a smaller percentage of the variance within the dataset. These uncorrelated PCs, or some subset of them, can be used in subsequent linear modeling of response variables while avoiding the problem of multicollinearity. Another objective of PCA is to reduce the dimensionality of the dataset. When a multivariate dataset consists of many correlated variables, the structure of the dataset can be explained with fewer transformed variables (PCs). A common rule of thumb for PCA is that the first *n* principal components with eigenvalues greater than one are the most important and explain most of the variance of a multivariate dataset with correlated variables (Jackson 1991).

In addition to PCA, factor analysis was also used to evaluate the relationships between sediment and porewater chemistry concentrations and response variables. Factor analysis is a subsequent orthogonal rotation of the axes conducted to create new variables called factors that might be more easily interpretable than PCs. Varimax rotation was used with the objective of having individual chemicals “load” (i.e., be strongly correlated with) a single factor and not load on any other factors. This is only achievable for multivariate datasets with “simple structure,” which is further described below.

PCA and factor analysis are subject to the assumptions of linear modeling, specifically that residuals are approximately normally distributed. Further, when variables that are measured in different scales and/or differ greatly in variance, are combined in PCA, they are commonly rescaled by standard normalization (or other methods) so as not give any one variable undue weight (Jackson 1991). These requirements were satisfied by conducting the PCA and factor rotation on a correlation matrix of natural logarithm transformed chemical concentrations. The PRINCOMP and FACTOR procedures in SAS© 9.4 were used to conduct the PCA and factor analysis.

6.1.2.3 Sediment

6.1.2.3.1 Principal Component Analysis of Sediment Chemistry

The objective of dose response modeling of the SQT data is to be able to predict where toxicity to the BIC might be expected to occur in the NBSA. Therefore, a PCA model was developed using the entire Phase III surface sediment dataset (n=254). Prior to conducting the PCA for sediment, an exploratory analysis was conducted to determine a subset of chemicals or summed totals that could be used to represent large chemical groups and also to simplify the analysis by reducing the variance introduced by chemicals that are unlikely to be causing the observed toxicity. Chemicals with a FOD less than 80% were excluded from the exploratory analyses due to the potential for adding additional unexplained variability. The details of this exploratory analysis are described in Appendix A. The following observations resulted in the subset of 21 chemicals/totals that were carried forward in the sediment chemistry PCA model:

- All dioxin and furan congeners are highly correlated; therefore, only 2,3,7,8-TCDD concentrations were used to represent this chemical group.
- PCB congeners are highly correlated; therefore, one total (total PCB congeners) was used to represent this chemical group.

- Individual PAH concentrations are correlated overall, but greater correlation was noted within PAHs grouped by alkylation and molecular weight. Therefore, three totals, total HMW, total LMW, and total alkylated PAHs were used to represent all PAH compounds.
- Exploratory analysis of metals concentrations provided evidence to exclude some individual metals that are likely mostly present at background concentrations due to their strong correlation with aluminum (beryllium, cobalt, iron, manganese, thallium, titanium, and vanadium). Other metals that were much less correlated with aluminum were carried forward in the PCA (antimony, arsenic, barium, cadmium, chromium, copper, lead, mercury, nickel, silver, and zinc).
- All 34 individual petroleum hydrocarbons were detected in surface sediments at frequencies ranging from 6 to 97% of samples. Although there were no sediment screening guidelines available for total petroleum hydrocarbons, they are potentially toxic to marine invertebrates (Hobbs et al. 2018) and were therefore included in the PCA. A summation, total petroleum hydrocarbons (C9-C40) was used to represent this class of chemicals.
- Because of differences in toxicity, no attempt was made to combine pesticides of different classes. Based on the observations previously described, such as detection frequency, exceedances of screening criteria, and correlation to effects measures, a subset of pesticides were selected for the PCA model. Total (2,4' and 4,4') DDx was used to represent all DDx isomers. Among chlordane chemicals, alpha- and gamma-chlordane were detected most frequently and often at values greater than screening guidelines; therefore, total alpha + gamma chlordane was used to represent chlordane chemicals. Hexachlorobenzene and dieldrin, both of which have high rates of detection in sediment and were fully detected in porewater, were also included in the PCA.
- No semivolatile compounds were included in the final PCA model. None were significantly correlated with toxicity and the low detection frequency (which would require substitution for non-detects because no values can be missing in PCA) would add uncertainty to the model.
- Two Phase III samples (not SQT samples) from locations 349 and 354 were collected in the area of the confined disposal facility. Chemical concentrations in the uncharacteristically sandy sediment samples from these locations were very low or non-detect adding additional variance to the model. Therefore, these samples were excluded from the final PCA as outliers. In addition, another sample near the confined disposal facility, at location 339, was excluded because of missing pesticide data. Concentrations of other chemicals were low at this location. This left a total of 251 Phase III surface sediment samples included in the sediment chemistry PCA.
- Detection frequencies were high among the remaining samples and 21 chemicals (>95%). One-half the ~~detection~~quantitation limit was substituted for any remaining non-detected results before the transformations were conducted.

The first principal component (PC1) explains 66.7% of the variance and the first three PCs explain a total of 83% of the variance of the sediment chemistry dataset (Figure 6-3). All remaining PCs have eigenvalues less than 1. All 21 chemicals are correlated to PC1. The second principal component (PC2), explaining 9.4% of the variance, is correlated with chlordane, dieldrin, 2,3,7,8-TCDD, and total PCBs. The third principal component (PC3) explaining 6.5% of the variance is correlated with PAHs. While PC1 explains most of the variance in overall chemical concentrations, PC2 and PC3 explain some relative differences in concentrations among the chemical groups. Because PCs are new variables that are linear

combinations of the original variables (i.e., chemical concentrations), the concentration data can be used to find the value, usually referred to as the score, of each PC for each sediment sample. Because PC1 represents overall chemistry concentrations, it is useful to view in map form to see patterns of elevated chemistry. Figure 6-4 presents the scores on PC1 of the Phase III surface sediment samples in the NBSA. Lower (more negative) PC1 scores indicate lower concentrations, while scores near 0 indicate median concentrations. Greater (more positive) scores indicate higher chemical concentrations.

6.1.2.3.2 Dose Response Modeling: Sediment

The objective of conducting the PCA was to create a set of variables, PCs, that represent sediment chemical concentrations (but are uncorrelated and therefore not subject to the problem of multicollinearity) to be used in the quantitative effects assessment. Linear modeling methods were used to evaluate the relationship between predictor variables (e.g., PCs, grain size, etc.) and response variables (e.g., 10-day mortality, number of taxa). Due to the nature of the response variables, two types of models were used: 1) linear/multivariate regression models—appropriate for continuous response variables (e.g., growth) and 2) binary response models—the appropriate model form for variables with only two possible outcomes (e.g., survival/mortality).

Linear regression models have the form:

$$R = \alpha + \beta_i x_i + \varepsilon \qquad \textbf{\textit{Equation 6-1}}$$

where,

R = response (e.g., growth, number of taxa)

α = intercept parameter

β_i = slope parameter of the *ith* predictor variable

x_i = *ith* predictor variable (e.g., PC1)

ε = random error

Competing linear regression models were compared using the adjusted coefficient of determination (R^2). The adjusted R^2 gives an indication of the ability of the model to explain the variance of the response data while penalizing the introduction of additional spurious predictors to the model. Models were fitted using the GLM procedure in SAS© 9.4. Parameters were added in a stepwise manner (step up and step down) and the R^2 and p-values for each parameter were compared using Type III (i.e., marginal) sums of squares. This process is detailed in tables in Appendix A.

Survival data from the 10- and 28-day *L. plumulosus* bioassays were evaluated using linear binary response models. Because of the natural or threshold mortality expected during the test as measured in the control, it is more convenient to model mortality rather than survival: (%Mortality = 100-%Survival). These models have the form:

$$\pi = C+(1-C)F(\alpha + \beta_i x_i) + \varepsilon \qquad \textbf{\textit{Equation 6-2}}$$

where,

π = probability of mortality

C = threshold mortality (i.e., mean laboratory control mortality)

F = a cumulative probability distribution function

α = intercept parameter

β_i = slope parameter of the *ith* predictor variable

x_i = *ith* predictor variable (e.g., PC1)

ε = random error

The cumulative distribution function (F) maps the range of the linear function ($-\infty,\infty$) onto the range of probability (0,1). Three different probability distributions were tested for each subset of model predictors: the logistic distribution, the standard normal distribution (called a probit model), and the Gompertz distribution (also called the extreme value model). Competing binary response models were compared using Akaike's Information Criteria (AIC). Given a set of candidate models, the preferred model is the one with the minimum AIC value (Akaike 1981). AIC rewards goodness of fit, but it also includes a penalty that is an increasing function of the number of predictors in the model.

For each response variable, the following predictors were tested: PC1, PC2, PC3, TOC, and percent fines. For BIC metrics, shear stress[6] was also tested. Predictor variables were added to each model in a stepwise fashion (i.e., step up and step down). In addition to the adjusted R^2 and AIC, the Type III marginal test statistic, computed for each parameter specified in the model, was examined to determine whether there is any contribution from the added predictor variable. Non-significant predictors (α = 0.05) were removed from the final models. Models were fitted using the PROBIT procedure in SAS© 9.4. Table 6-14 summarizes the significant (α = 0.05) and best fitting models for each toxicity endpoint and BIC metric tested. Full details of model fitting and graphs of fitted models are presented in Appendix A.

For the 10-day test, the best fitting model for amphipod mortality was a Probit model with PC1 as the sole predictor variable (Figure 6-5). Based on this model, mortality in all Phase III sediments was calculated. Mortality was transformed back to survival (%Survival = 100-%Mortality) and samples for which survival is predicted to be less than 80% and/or 60% of the control survival were identified on a map (Figure 6-6). These locations are labelled with their abbreviated location identifiers (i.e., station identifier). Based on the model, acute toxicity is expected in samples with a PC1 score of 5 or greater for which there are a total of 21 (8.2% of all Phase III samples) in the NBSA, three in the north geographic area, and 18 in the south geographic area.

For the 28-day test, the best fitting model for amphipod mortality was a Gompertz model with both PC1 and PC2 as significant predictors (Figure 6-7). It is observed that 28-day mortality is much more variable than 10-day mortality and therefore the predictability of the model is less certain. However, based on the observed results, the model correctly predicts toxicity (or lack thereof) 80% of the time (see Appendix A for the analysis of model predictability). Similarly, as was done for the 10-day model, mortality in all Phase III sediments was calculated and transformed back to survival. Samples for which 28-day survival is predicted to be less than 80% and/or 60% of the control survival were identified on a map (Figure 6-8). Chronic toxicity is predicted in 45 (17.7%) of Phase III samples: 11 are located in the north geographic

[6] Shear stress is defined as the daily maximum grain stress in dynes per square centimeter predicted by the hydrodynamic model (1995-2013) of the NBSA in the model grid cell in which the SQT sample was located.

area mostly in the same area where acute toxicity was predicted; 5 are spread out in the central geographic area; and the remaining 29 samples are in the southern geographic area mostly in the same areas where acute toxicity was predicted.

PC1 (and no other PCs) was a significant negative predictor for 28-day growth (R^2 = 0.19). Mean growth in the laboratory control sample was 0.045 milligram (mg)/organism/day. Only one SQT sample met or exceeded this value (Table 6-4). The SQT sample with the lowest chemical concentrations, from location 155, had observed growth of 0.035 mg/organism/day. As previously discussed, the sublethal endpoints are subject to considerably more variability that is unrelated to chemical stressors. Kennedy et al. (2009) found significant variability in control sediments for growth with a CV of 56% as well as significant variability in an interlaboratory comparison of growth. Eickhoff et al. (2014) found considerable variability in mean growth in five control sediments from the same estuary ranging from 0.025 to 0.064 mg/organism/day. The only test acceptability criterion for control growth is that it is measurable (USEPA 2001), setting a low bar for this endpoint. Growth was measurable in all but two of the SQT samples that exhibited 100% mortality at 28 days (Table 6-4). It is unknown what growth rate could be achieved in the NBSA sediments in the absence of chemical stressors. However, based on the results of sample 155, it is more likely to be in the range of 0.035 mg/organism/day or less. At the median PC1 score of 0, the regression model predicts mean growth of 0.03 mg/organism/day, a 35% reduction compared to control but within the range of growth observed in control sediments by Eickhoff et al. (2014). At a PC1 score of 5, the threshold for prediction of acute toxicity, the regression model predicts mean growth of 0.02 mg/organism/day, a reduction of 57% from control growth.

The reproductive endpoint was not significantly associated with any of the predictors tested, which is not surprising due to the variability in this endpoint. Reproduction in the laboratory control (mean = 1.97 young per surviving adult) was subject to significant replicate variability where it ranged from 0.33 to 4 young per surviving adult (CV = 73%). Among the SQT samples, reproduction ranged from 0 to 1.93 (mean = 0.68 young per surviving adult) (Table 6-4). As previously discussed, the reproductive endpoint is subject to considerable variability unrelated to chemical stressors. Eickhoff et al. (2014) observed mean reproduction in five control samples from the same estuary ranging from 0.8 to 8.4 young per surviving adult with an average replicate CV of 79%. Kennedy et al. (2009) found significant variability in control sediment for reproduction with a CV of 63%, as well as significant variability in an interlaboratory comparison of reproduction. Reproduction is also known to be highly dependent on grain size (McDonald et al. 2010).

Among benthic metrics, only density was significantly negatively correlated with PC1 (or any other PCs) (R^2 = 0.17). Inferences about density vs. chemical concentrations are subject to uncertainty because chemical stressors can result in both increases and decreases in this metric (Weisberg et al. 1998). The number of taxa was most correlated with grain size (R^2 = 0.29) with an increasing percentage of silt and clay resulting in a decline in number of taxa.

6.1.2.3.3 Factor Analysis of Sediment Chemistry

In addition to PCA, factor analysis was conducted on the sediment chemistry data to determine if individual chemicals or chemical classes could be attributed to the observed toxicity. Based on the outcome of the PCA, it was hypothesized that three factors would explain the variance of the multivariate dataset. Varimax rotation resulted in the factor loadings shown on Figure 6-9. The objective of varimax

rotation is to have each chemical load on one factor and no other. This was not fully achievable for the NBSA sediment chemistry dataset; however, some chemicals loaded more strongly on individual factors. Metals were most strongly correlated to Factor 1 (Figure 6-9). Pesticides, PCBs, and 2,3,7,8-TCDD were most strongly correlated to Factor 2. However, the pesticides, total DDx and hexachlorobenzene also had significant loading on Factor 1 with metals. PAHs and total petroleum hydrocarbons were most strongly correlated to Factor 3. However, alkylated PAHs and total petroleum hydrocarbons also had significant loading on Factor 1. Dose response modeling was conducted using these factors in a similar manner as was described for the PCA (Table 6-15).

For 10-day mortality, both Factors 1 and 3 were significantly and positively correlated with mortality (Table 6-15). For 28-day mortality, both Factors 1 and 2 were significantly and positively correlated with mortality. No factor models were significant for growth or reproduction. These findings are consistent with the findings in the PC models where PC1, representing elevated chemistry for all chemicals, was a significant predictor of both 10- and 28-day toxicity and PC2 was also significant for 28-day mortality.

6.1.2.3.4 Sediment Conclusions

The PCA, factor analysis, and dose response modeling provide evidence that chemical concentrations in sediments are associated with toxicity to benthic invertebrates. The PCA-based dose response models can be used to reasonably predict where toxicity has the potential to occur in the NBSA as shown on Figures 6-6 and 6-8. The factor-based dose response models confirm that multiple chemical mixtures are associated with the observed toxicity. Major findings include:

- The PCA of sediment chemistry concentrations indicates that most of the variance is explained by the first three PCs with PC1 explaining the most variance since it is representative of overall concentration and most of the chemicals are correlated with each other (Figure 6-3).
- The 10-day toxicity testing resulted in only two toxic stations with low survival while all other stations had very high survival. These two stations also had the highest concentrations of many chemicals in sediment and porewater (Table 6-10) resulting in ~~a strong~~an association between acute toxicity and PC1. The model predicts acute toxicity at a PC1 score of 5; however, a threshold for acute toxicity could occur between a PC1 of 3.5 (i.e., the highest PC1 score for a non-toxic sample) and 5.6 (i.e., the lowest PC1 score for a toxic sample).The uncertainty with respect to estimating a threshold for acute toxicity is discussed in Section 6.4.
- The 28-day survival results were subject to considerably more variability and the relationship to chemistry, while significant, is not as strong as it is for the 10-day tests. Both PC1 and PC2 are associated with 28-day survival (Figure 6-7). Two stations, 142 and 158, that showed significant departure from the model (Figure 6-7) also had significant within replicate variability for this endpoint (Appendix A).

6.1.2.4 Porewater

PCA was conducted using the porewater chemistry data. To be consistent with the sediment chemistry PCA where possible, the same subset of chemicals was used in the porewater PCA:

- Porewater concentrations of 2,3,7,8-TCDD, detected in all porewater samples, were used to represent all dioxin and furan congeners.
- Most PCB congeners had high rates of detection in porewater with 111 congeners/co-elutions having a 100% detection rate. Total PCB congeners was used to represent all PCB congeners in the PCA.
- Individual PAH concentrations are represented by two totals, total PAHs and total alkylated PAHs, both 100% detection rates. Total LMW PAHs had a FOD rate of only 50% in porewater and were therefore not evaluated separately from HMW PAHs as was done for sediment.
- Except for mercury, only 29 samples were analyzed for metals due to a broken dialysis bag for the sample from station 155. Of the remaining samples, most metals had very low FOD: antimony (3%), arsenic (31%), cadmium (0%), copper (17%), lead (21%), nickel (0%), silver (0%), and zinc (10%). Therefore, only mercury, which was detected in 29 of 30 samples, was included in the porewater PCA model. One-half the ~~detection~~quantitation limit was substituted for the non-detect.
- Total (2,4' and 4,4') DDx was used to represent all DDx isomers and was fully detected in porewater. Among chlordane chemicals, alpha- and gamma-chlordane were fully detected with concentrations as much as 10 times greater than other chlordane chemicals; therefore, total alpha + gamma chlordane was used to represent chlordane chemicals. Hexachlorobenzene and dieldrin, both fully detected in porewater, were also included in the PCA.
- Semivolatiles and petroleum hydrocarbons were not analyzed in porewater.

A total of nine chemicals was included in the porewater PCA (Figure 6-10). The first principal component (PC1) explains 51% of the variance and the first three PCs explain a total of 80% of the variance of the sediment chemistry dataset (Figure 6-10). All remaining PCs have eigenvalues less than 1. All but mercury and total alpha + gamma chlordane are correlated to PC1. The second principal component (PC2), explaining 19% of the variance, is correlated with mercury and total alpha + gamma chlordane. The third principal component (PC3), explaining 10% of the variance is correlated with mercury and PAHs. While PC1 explains most of the variance in overall chemical concentrations, PC2 and PC3 explain some relative differences in concentrations among the chemical groups.

Based on the PCA, three factors were evaluated for porewater (Figure 6-11) using varimax rotation with the objective of having chemicals strongly load (i.e., be correlated with) a single factor and no other factors. However, this was not achieved for the porewater chemistry dataset. Total PCB congeners, 2,3,7,8-TCDD, hexachlorobenzene and dieldrin are most strongly correlated with Factor 1 (45% of the variance). PAHs are most strongly correlated to Factor 2 (35% of the variance) and mercury is most strongly correlated with Factor 3 (20% of the variance). Total DDx and alpha + gamma chlordane are correlated with all factors; although, in opposite directions for Factors 2 and 3. Total DDx is positively correlated to Factor 2 and negatively correlated to Factor 3, while the opposite is observed for alpha + gamma chlordane.

Both PCs and factors were evaluated in dose response modeling of porewater chemistry and toxicity (Table 6-16). The modeling methods were the same as those described in Section 6.1.2.3.2 for sediment. Only PCs and factors were tested as predictors for toxicity. PC1 (and no other PCs) is significantly and positively associated with 10-day mortality (Table 6-16; Figure 6-12). Both factors 1 and 2 were significantly and positively correlated with 10-day mortality (Table 6-16); however, combining them in one

model resulted in unstable parameter estimates. This indicates that both factors may be giving similar or redundant information, which makes sense given that both of the two acutely toxic stations, 151 and 154, have relatively high scores on both factors. Therefore, the chemicals (or some subset of them) associated with PC1 and both factors 1 and 2 (i.e., all except mercury and total alpha + gamma chlordane) are potentially contributing to the observed acute toxicity.

Porewater PCs were also evaluated as predictors of BIC metrics (Appendix A) and were not found to be significant predictors of any BIC metrics.

6.1.2.5 Non-metric Multidimensional Scaling

Benthic metrics are useful indicators of BIC health, but some information is lost in the analysis of metrics only. Therefore, community ordination was also evaluated with respect to sediment chemistry. Ordination refers to a variety of techniques used to arrange benthic samples in relation to one or more coordinate axes and to each other to provide information about their ecological similarities. Community abundance data collected over environmental gradients are typically non-linear and are better suited to non-linear methods of analysis such as non-metric multidimensional scaling (NMDS) of a Bray-Curtis dissimilarity matrix (Ludwig and Reynolds 1988). Like PCA, NMDS is a method used to reduce the dimensionality of the community data, the result of which is that each BIC sample can be scored in the new smaller coordinate space. These scores can be plotted on graphs where samples near each other are judged more similar than those further away on the graph. Spatial patterns may be discernible if the BIC varies based on geographic or geomorphic location. The resulting dimensions can be regressed against other variables such as individual chemicals, PCs, or factors to evaluate effects of stressors on changes in community structure.

Figure 6-14 illustrates the results of the NMDS evaluation. Goodness of fit criteria indicate that the community structure can be represented in two dimensions (see Appendix A). The space between samples can be thought of as distances that increase with dissimilarity. For example, stations 148 and 160 have no species in common and exhibit the maximum distance between any two stations on the graph Figure 6-14a). In this type of analysis, the dimensions are not always easily interpretable; however, dimension 2 is correlated with number of taxa. There are no obvious geographical trends (Figure 6-14a). Other spatial trends based on geomorphic unit or USEPA Subunit were also not apparent (Appendix A). Linear egression modeling was conducted to determine if the sediment chemistry PC1 was a predictor of community structure (Figure 6-14b). There is no significant relationship between PC1 and dimension 1 ($p=0.26$) nor between PC1 and dimension 2 ($p=0.38$). It might appear from the graph (Figure 6-14b) that a relationship between PC1 and dimension 2 would be more apparent if the two toxic stations (151 and 154) were removed from the analysis; however, the p-value of this model is still not significant ($p=0.09$). Other predictors for BIC dimensions were tested: PC2, PC3, TOC, and percent fines (Appendix A). The only finding was that percent fines (Figure 6-14c) is a significant predictor for dimension 2 ($p=0.004$; $R^2 = 0.26$), which is not surprising given that percent fines was also the best predictor for number of taxa.

In summary, there is no evidence of geographic spatial patterns among the benthic community structure and no apparent relationship to sediment chemistry. Sediment grain size is the best predictor of patterns in community structure.

6.1.2.6 Summary of Quantitative Effects Assessment

The quantitative effects assessment demonstrated the following conclusions.

Acute toxicity is significantly correlated with chemical concentrations in sediment and porewater. The PCA evaluations demonstrated that measures of overall chemical concentrations as represented by the first PC in both the sediment and porewater PCA are significantly correlated with 10-day amphipod mortality and based on this relationship, areas where acute toxicity might occur can be reasonably predicted based on the Phase III sediment chemistry (Figure 6-6). The factor analyses demonstrated that toxicity is associated with a mixture of multiple chemicals: metals, pesticides, PCBs, PAHs, and dioxins/furans. No single chemical or chemical group in the evaluation was shown to be unrelated to toxicity. It is important to note that because of the high correlation among chemical concentrations, these findings cannot prove that all individual chemicals evaluated are causing toxicity. Acute toxicity was only observed at two of the 30 SQT stations in the NBSA. Those two stations happened to have the highest concentrations in both sediment and porewater of most chemicals including those included in the multivariate analysis (Table 6-10).

Chronic toxicity is also significantly correlated with chemical concentrations in sediment and porewater as demonstrated by the PCA models. And similarly factor analysis demonstrated that toxicity is associated with a mixture of multiple chemicals: metals, pesticides, PCBs, PAHs and dioxins/furans. Based on models of 28-day mortality, areas where chronic toxicity could potentially occur can be reasonably predicted based on the Phase III sediment chemistry (Figure 6-8).

For chronic sublethal endpoints, growth but not reproduction is significantly negatively correlated with chemical concentrations; although, there is significant variability in the sublethal responses.

With the exception of density, none of the BIC metrics or community ordination measures were significantly correlated with chemical concentrations. PC1 is a significant predictor of density ($p=0.026$; $R^2 = 0.17$); however, because density can either increase or decrease as a result of degradation (Weisberg et al.1998), this somewhat weak relationship does not demonstrate chemical impact to BIC.

6.1.3 Integrated Benthic Community Risk Characterization

Table 6-17 summarizes the lines of evidence of the SQT Assessment based on the individual evaluations of BIC, toxicity, and sediment and porewater data as well as the quantitative effects assessment. The dose response modeling and comparisons to Jamaica Bay reference indicate no apparent impact to the BIC due to chemical stressors. Multivariate analyses indicate that acute and chronic toxicity to amphipods are associated with chemical mixtures in sediment and porewater. Toxicity occurred in samples with elevated concentrations of multiple chemicals: dioxins/furans, PCBs, pesticides (particularly total DDx, dieldrin, chlordane, and hexachlorbenzene), PAHs (both parent an alkylated), petroleum hydrocarbons, and most inorganics (e.g., arsenic, cadmium, chromium, copper, lead, mercury, and zinc). Areas where the dose response models predict potential toxicity are of limited extent in the NBSA and are primarily located in the southwestern subtidal flat and in a smaller area in the north in the vicinity of historically disturbed sediments.

6.2 Benthic Invertebrate Tissue Assessment

The tissue assessment was conducted for epibenthic shellfish including the softshell clam and blue crab. In addition, the tissue data generated from the polychaete worm bioaccumulation tests were also evaluated as another infaunal invertebrate LOE. The COPECs evaluated for the benthic invertebrate tissue assessment are listed in Table 3-5 and include: 2,3,7,8-TCDD, total PCBs (209 congeners), pesticides (total chlordane, total DDx, dieldrin, and hexachlorobenzene), PAHs, and inorganics/metals (arsenic, cadmium, chromium, copper, lead, mercury/methylmercury, nickel, selenium, silver, and zinc).

Tissue data EPCs for each species from the 2014 to 2016 BERA sampling program were compared to whole-body tissue TRVs to calculate HQs. The risk characterization presents the results of this assessment and summarizes the uncertainties associated with the tissue assessment.

6.2.1 Exposure

Tissue EPCs were calculated as UCLs using all available whole-body tissue data (i.e., individual and composite; calculated whole-body samples) for each species. The UCLs were calculated using USEPA's ProUCL® statistical package (Version 5.1) (USEPA 2015b) as described in Section 4.2.6. If a dataset contained fewer than six detected concentrations, a UCL was not calculated; instead, the maximum concentration was used as the EPC. Table 4-11 presents the sample locations in each assessment zone for benthic invertebrate (i.e., polychaete worm, blue crab, and softshell clam) tissue. The COPEC summary concentrations for invertebrates are provided in ~~Table~~Tables 4-~~12,~~7a-c and the selected EPCs for invertebrate tissue samples by zone are presented in Appendix C.

6.2.2 Effects

Benthic invertebrate tissue-based TRVs for the COPECs identified in Table 3-5 are provided in Table 6-18, and discussed in detail in Appendix D. The selected TRVs were primarily derived from the LPRSA BERA TRV database (Windward 2019~~).~~) and are consistent with the Final LPRSA BERA TRVs for COPECs common to both areas (NBSA and LPRSA). For the LPRSA BERA, Windward (2019) conducted a literature search for relevant toxicological studies, and derived acceptable NOAELs and LOAELs. The lowest acceptable LOAEL and highest NOAEL below the LOAEL from the same study based on marine organisms were chosen. If marine TRVs were not available, then the freshwater-based TRV selected by LPRSA BERA was used. ~~Several~~Two COPECs being evaluated for invertebrate tissue assessment in this BERA~~ were not evaluated in the LPRSA BERA, including total PAHs~~, total chlordane~~,~~ and hexachlorobenzene~~.~~, were not evaluated in the LPRSA BERA. The selected NBSA TRVs for these additional COPECs are further discussed in Appendix D.

~~Separate TRVs were selected for 1) blue crab and polychaete worm and 2) softshell clam for 2,3,7,8-TCDD and total PCBs. To evaluate softshell clams exposed to 2,3,7,8-TCDD, a NOAEL and LOAEL TRV from Cooper and Wintermyer (2009) based on reproduction in oyster and softshell clam was selected. To evaluate blue crab and polychaete worms exposed to 2,3,7,8-TCDD, a NOAEL and LOAEL TRV from Ashley et al. (1996) based on survival of pink shrimp was selected. Softshell clams exposed to total PCBs were evaluated using a NOAEL and LOAEL TRV from Chu et al. (2000) and Chu et al. (2003) based on reproduction in Eastern oyster. Blue crab and polychaete worm exposed to total PCBs were evaluated~~

~~using a NOAEL and LOAEL TRV from Hansen et al. (1974) based on survival, growth, and reproduction in 10 species.~~

In addition to the above selected TRVs for the NBSA BERA, TRVs derived by the USEPA for the LPR FFS (USEPA 2014) were also assessed and are included in the TRV summary table (Table 6-18). LPR FFS benthic invertebrate tissue TRVs are available for 2,3,7,8-TCDD, total PCBs, total DDx, dieldrin, LMW PAH, HMW PAH, copper, lead, and methylmercury. A key difference in the TRV application for invertebrates in the LPR FFS is the use of oyster (i.e., bivalve) TRVs to assess potential risks to blue crab for 2,3,7,8-TCDD (derived from Wintermyer and Cooper 2003) and PCBs (derived from Chu et al. 2000; Chu et al. 2003). This is a highly conservative and uncertain TRV application~~,~~ given the very different life histories and exposure potential between sessile filter-feeding oysters and mobile blue crab~~.~~, but was used for risk bounding given the limited TRVs available for crustaceans for meaningful endpoints other than survival.

The NBSA total DDx TRV for benthic invertebrates was a SSD-derived 5th percentile value based on survival, growth, and reproduction studies in six species (NOAEL = 0.011 mg/kg and LOAEL = 0.11 mg/kg). This SSD-derived value was based on a gamma distribution. An additional alternative total DDx TRV was also developed by Windward (2019) using the same SSD dataset, but the TRV was derived from a beta general distribution (NOAEL = 0.001 mg/kg and LOAEL = 0.01 mg.kg). Both sets of total DDx TRVs for invertebrates are assessed in this NBSA BERA.

6.2.3 Risk Characterization

This section presents the comparison of benthic invertebrate tissue EPCs to NBSA-selected and LPR FFS TRVs to calculate HQs for each COPEC and species. Benthic invertebrate tissue (i.e., worm, crab, and clam) NOAEL and LOAEL HQs are presented in Table 6-19a (NBSA-wide), Table 6-19b (North Zone), Table 6-19c (Southeast Zone), and Table 6-19d (Southwest Zone). The HQs are based on UCLs, or maximum concentrations if there were fewer than six detected values. Polychaete worms were not evaluated by assessment zone due to low sample sizes (i.e., total of eight samples Bay-wide from the bioaccumulation tests).

A summary of COPECs that exceed their TRVs (i.e., HQs that equal or exceed 1) is provided in Table 6-20.

Table 6-20. Summary of TRV Exceedances for Benthic Invertebrate Tissue

~~Constituents~~	~~NBSA and LPR FFS NOAEL and LOAEL-based HQs ≥ 1.0~~																	
	~~Polychaete Worm~~		~~Softshell Clam~~								~~Blue Crab~~							
	~~NBSA-wide~~		~~NBSA-wide~~		~~North~~		~~South east~~		~~South west~~		~~NBSA-wide~~		~~North~~		~~South east~~		~~South west~~	
	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~
~~Dioxins/Furans~~			-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~2,3,7,8-TCDD~~		●	●	●	●	●	●	●	●	●		●		●		●		●

~~**PCBs**~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~Total PCBs~~		●	●	●	●	●	●	●	●	●		●	⊖	●		●		●
~~**Pesticides**~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~Total DDx (2,4 & 4,4)~~											⊖	⊖	⊖	⊖	⊖	⊖	●	●
~~Dieldrin~~	⊖	⊖									⊖	⊖	●	●	⊖	⊖	⊖	⊖
~~Hexachlorobenzene~~	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—
~~**PAHs**~~			-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~Total HMW PAH~~	⊖	⊖	●	●	●	●	●	●	●	●	⊖	⊖	⊖	⊖	⊖	⊖		
~~Total LMW PAH~~									⊖	⊖					⊖	⊖		
~~Total PAH~~	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—
~~**Inorganics**~~			-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~Arsenic~~	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—
~~Cadmium~~	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—	⊖	—
~~Chromium~~		—	⊖	—	⊖	—	●	—	●	—		—		—		—		—
~~Copper~~			●	●	⊖	⊖	●	●	●	●	●	●	●	●	●	●	●	●
~~Lead~~	⊖	⊖	⊖	●	⊖	●	⊖	●	⊖	●	⊖	⊖	⊖	⊖	⊖	⊖	⊖	⊖
~~Mercury~~			●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●
~~Methylmercury~~							⊖	⊖	⊖	⊖	●	●	●	●	●	●	●	●
~~Nickel~~	⊖	—	●	—	●	—	●	—	●	—	⊖	—	⊖	—	⊖	—	⊖	—
~~Selenium~~	⊖	—	●	—	●	—	●	—	●	—	●	—	●	—	●	—	●	—
~~Silver~~		—		—		—	●	—		—	●	—	●	—	●	—	●	—
~~Zinc~~	●	—	●	—	●	—	●	—	●	—	●	—	●	—	●	—	●	—

Constituents	NBSA and FFS NOAEL and LOAEL-based HQs ≥ 1.0																	
	Polychaete Worm		Blue Crab								Softshell Clam							
	NBSA-wide		NBSA-wide		North		Southeast		Southwest		NBSA-wide		North		Southeast		Southwest	
	NBSA	FFS	NBSA	FFS	NBSA	FFS	NBSA	FFS	NBSA	FFS	NBSA	FFS	NBSA	FFS	NBSA	FFS	NBSA	FFS
Dioxins/Furans																		
2,3,7,8-TCDD		●		●		●		●		●		●		●		●		●
Polychlorinated Biphenyls																		
Total PCBs	○	●	○	●	○	●	○	●	○	●	○	●	○	●	○	●	○	●
Pesticides																		
Total DDx (2,4 & 4,4)			●	○	●	○	○	○	●	●	○				○		○	
Total DDx (2,4 & 4,4) Alternative TRV	○	--	●	--	●	--	●	--	●	--	●	--	●	--	●	--	●	--
Dieldrin		○		○	○	●		○		○								
Total Chlordane		--		--		--		--		--		--		--		--		--
Hexachlorobenzene		--		--		--		--		--		--		--		--		--
Polycyclic Aromatic Hydrocarbons																		
Total HMW PAH	○	○	○	○		○		○			●	●		○		○		○
Total LMW PAH								○										○
Inorganics																		
Arsenic	●	--	●	--	●	--	●	--	●	--	●	--	●	--	●	--	●	--
Cadmium	○	--	●	--	●	--	●	--	●	--	○	--	○	--	●	--	●	--
Chromium		--		--		--		--		--	●	--	●	--	●	--	●	--
Copper	--		--	●	--	●	--	●	--	●	--	●	--	○	--	●	--	●
Lead		○		○		○		○		○	○	●	○	●	○	●	●	●
Mercury			●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●
Methylmercury			●	●	●	●	●	●	●	●					○	○	○	○
Nickel	○	--	○	--	○	--	○	--	○	--	●	--	●	--	●	--	●	--
Selenium	○	--	●	--	●	--	●	--	●	--	●	--	●	--	●	--	●	--
Silver		--	●	--	●	--	●	--	●	--		--		--	●	--		--
Zinc	○	--	○	--	○	--	○	--	○	--	○	--	○	--	○	--	●	--

Notes: ● = indicates LOAEL HQ ≥ 1; ○ = indicates NOAEL HQ ≥ 1; - = not evaluated in the LPR FFS; COPECs evaluated that did not have any exceedances are not shown.

~~Both NOAEL and~~ LOAEL HQs ≥ 1 were calculated NBSA-wide for softshell clam for total DDx (alternative TRV only), total HMW PAH, and several inorganics (arsenic, chromium, mercury, nickel, and selenium) using the NBSA TRVs, and 2,3,7,8-TCDD, total PCBs, total HMW PAHs, and several ~~metals~~inorganics (copper, ~~mercury, nickel, selenium, and zinc). These were comparable for 2,3,7,8-TCDD, total PCBs, PAHs~~lead, and mercury) using ~~both the NBSA and~~ LPR FFS TRVs. ~~For blue crab, both NOAEL and~~ Similar results were found in each of the zones (see Table 6-20).

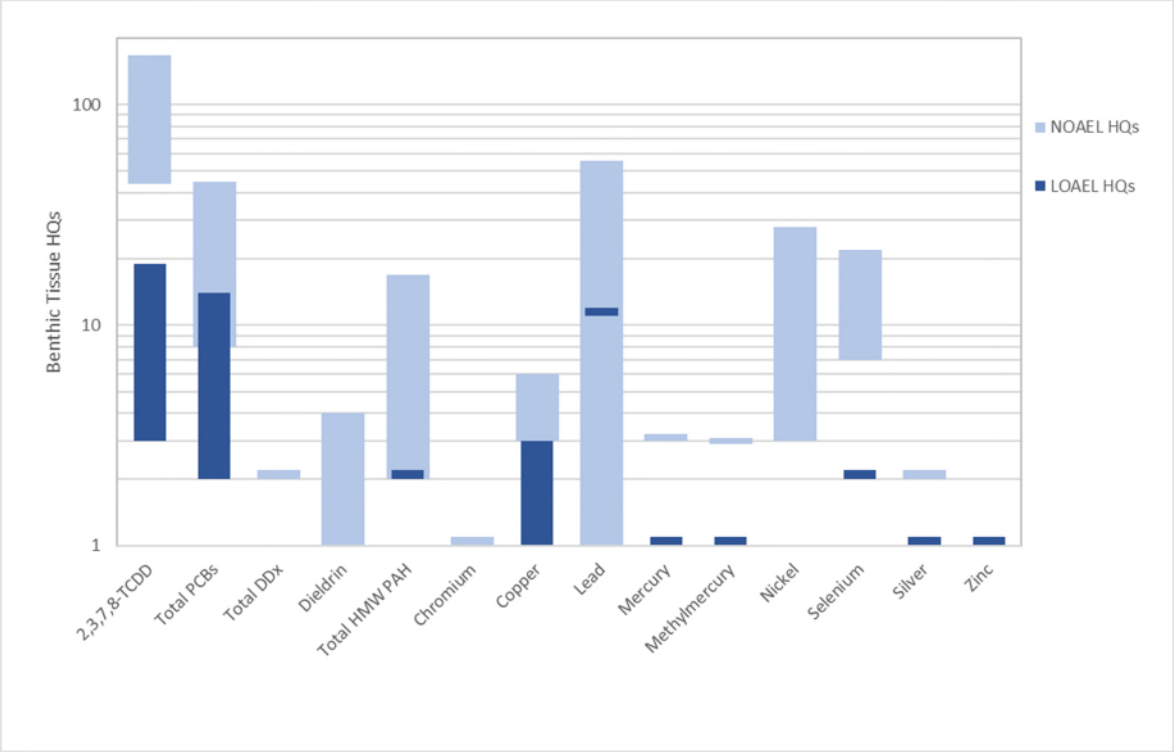
LOAEL HQs ≥ 1 were calculated ~~for dieldrin in the North Zone, total DDx in the Southwest Zone, and~~ NBSA-wide for blue crab for total DDx (both primary and alternative TRVs) and several inorganics (arsenic, cadmium, mercury, methylmercury, selenium, and silver) using the NBSA TRVs, and 2,3,7,8-TCDD, total PCBs, and several inorganics (copper, mercury, and methylmercury) for ~~several inorganics NBSA-wide using NBSA TRVs. In addition, blue crab data exceed the~~ LPR FFS ~~oyster TRV for 2,3,7,8-TCDD and total PCBs, but not the NBSA TRV.~~TRVs. Similar results were found in each of the assessment zones (see Table 6-20).

The polychaete worm bioaccumulation data ~~exceeded several NOAEL TRVs, but only exceeded~~ LOAEL HQs were ≥ 1 for 2,3,7,8-TCDD and total ~~PCB LOAELs~~PCBs using ~~the~~ LPR FFS TRVs, and ~~HQs equal~~

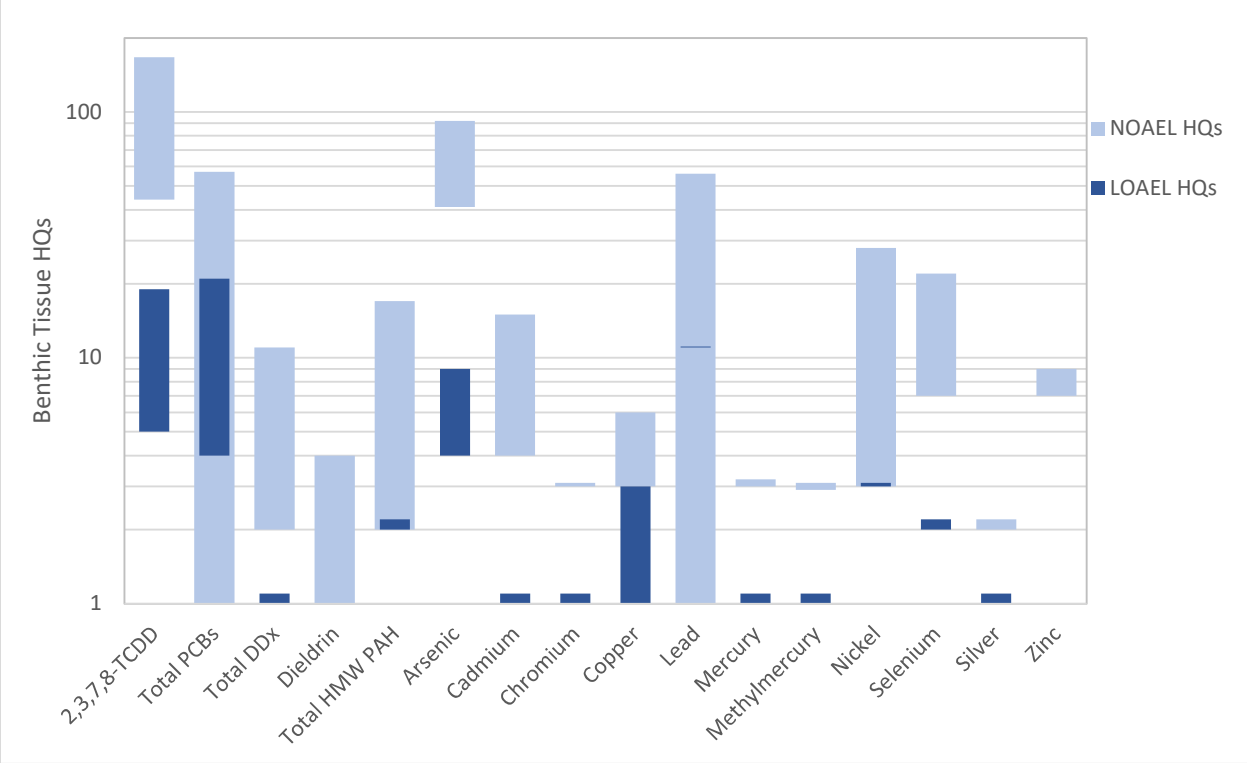
~~1~~only arsenic for ~~zinc using the~~ NBSA ~~TRV~~TRVs. The limited polychaete tissue data generated from the laboratory exposures are compared to tissue TRVs in order to provide perspectives on the overall potential for invertebrate risks. They may be considered a secondary line of evidence for risk, but the primary risk conclusions for these organisms are made in the SQT assessment from toxicity and community-based analyses. Polychaete survival ranged from 92 to 100% during the 28-day bioaccumulation study for the NBSA indicating the insensitivity of these organisms to the NBSA COPECs. In addition, a site-specific study in the NBSA system of polychaete bioaccumulation and toxicity conducted in the LPR (i.e., Iannuzzi et al. 2008) has shown that elevated concentrations of similar COPECs as in the NBSA but at higher concentrations, do not affect the growth or survival of these organisms.

The range of HQs for the aggregate invertebrate tissue samples (i.e., all species) with HQs ≥ 1 are depicted in Figure 6-15. ~~In general, the magnitude of LOAEL HQs ≥ 1 for all invertebrates using the NBSA TRVs were low (i.e., between 1 and 4), suggesting limited potential for risk. For the LPR FFS TRVs, higher invertebrate LOAEL HQ values were calculated for 2,3,7,8-TCDD (5 to 19), total PCBs (2 to 14), and a lead LOAEL HQ of 11 for softshell clam.~~

Figure 6-15. Magnitude of NBSA-wide NOAEL and LOAEL HQs ≥ 1 for Invertebrate Tissue



Assessment



6.3 Additional Lines of Evidence for Benthic Invertebrates

Additional LOEs for bivalves include site-specific toxicity testing conducted with transplanted oysters in the NBSA (Wintermyer and Cooper 2003). This study evaluated the bioaccumulation of dioxins/furans and PCBs and concurrent reproductive success of transplanted oysters in Newark Bay and other study sites in the NY/NJ Harbor Estuary and was the basis of the LPR FFS tissue TRVs for those COPECs. The study found diminished reproductive success in oysters deployed in the Bay. While it was focused on dioxins/furans and PCBs, the authors of this study did not take into account the various other contaminants present in the study system.

6.4 Uncertainties Assessment

The following is an uncertainties assessment for the invertebrate risk assessment. Both quantitative and qualitative characterizations are provided for the SQT and tissue assessments presented in Sections 6.1 and 6.2, respectively. The objective of the uncertainty assessment is to provide perspective on the likely accuracy of the overall risk conclusions that are presented in Section 6.5.

One overarching uncertainty that is relevant to both the SQT and invertebrate tissue assessments is the fact that a formal risk analysis was not conducted for the regularly dredged deep water channels in the NBSA. This was a decision made during the planning of the BERA process due to the changing conditions in these areas from regular navigational dredging activities performed by the USACE, which regularly removes sediments from these channels. Ten composite sediment samples were collected in

the regularly dredged deep water channels. Summary statistics for these samples are presented in Table 4-12. Mean and UCL concentrations for most of the COPECs including 2,3,7,8-TCDD, total PCBs, PAHs, some pesticides, and most metals are generally lower in the channel sediments than those in the other ecological risk assessment zones. Exceptions include total DDx, chlordane, mercury/methylmercury, silver, and zinc for which the average and UCL concentrations are within the range of the other zones. Given that the concentrations of COPECs in the channels are below or within the range of those found in the other zones, the potential for risks to invertebrate receptors that are exposed in the channels are bounded by the risk estimates for the other zones presented in this BERA.

6.4.1 SQT Uncertainties

There are several uncertainties associated with the SQT assessment related to the following issues: weighting and scoring in the SQT evaluation; estimation of porewater concentrations; prediction of acute toxicity; variability in chronic toxicity endpoints; reference comparisons and the representativeness of the 30 SQT samples. These are each discussed in the following paragraphs.

The scoring in the SQT evaluation used equal weights for each category of the triad: BIC, toxicity, and sediment chemistry. Alternative scoring methods have been proposed in the literature that give less weight to data with more uncertainty, in particular, the sediment chemistry category because it is generally based on a comparison to screening values (i.e., T20/T50 and ERL/ERM) that have not been demonstrated to accurately predict toxicity as they were derived from field sediments with chemical mixtures. In this assessment, the sediment chemistry score is based on an equal weighting of sediment and porewater chemistry. While the sediment chemistry portion of the score is based on a comparison to screening values, the porewater chemistry portion of the score is based on exceedances of promulgated water quality criteria, which may be a better indicator of exposure and potential impact for benthic communities. Therefore, to assess this uncertainty, the scoring was recalculated, to give porewater chemistry the full weight of the sediment chemistry category score (i.e., sediment chemistry is ignored in the scoring). The result of the scoring change is that seven of the low impact stations become no impact stations for a total of 17 no impact stations. The two stations categorized as medium impact (i.e., stations 151 and 154) remain so. The remaining 11 stations keep their low impact classification. This alternative scoring is perhaps a better characterization of risk given the paucity of observed impacts to BIC and the low levels of observed acute toxicity. Another alternative scoring scenario was evaluated that assigned a score of 1 to any survival value less than 80% of control survival, regardless of statistical significance. Variability in the replicate data could result in MDD that would not identify samples as toxic when survival is less than 80% of control (see Attachment A-2 for estimates of MDDs for each endpoint). The outcome of this scoring is nine no impact stations and 19 low impact stations. The medium impact stations maintain their categorization. The details of the alternative scoring evaluations are presented in Attachment A-5 of Appendix A.

While it is expected that porewater concentrations would provide better information about exposure, it is important to note the uncertainty in the measurement of porewater chemistry. Concentrations of organic compounds in porewater were obtained by ex-situ exposure of polyethylene (PE) and polyoxymethylene (POM) passive samplers to sediments in a laboratory-controlled setting. PE passive samplers loaded with performance reference compounds (PRCs) were used to determine the freely dissolved porewater concentrations of PCBs, PAHs, and pesticides. POM passive samplers (without PRCs) were used to

measure freely dissolved concentrations of dioxins and furans. Following exposure to sediment, the PE and POM samplers were transferred to the analytical laboratories where the target organic compounds were extracted in solvent (acetone/hexane (1:1 by volume)) and the concentrations in final extracts were measured. Porewater concentrations were estimated using the laboratory-reported concentrations in the PE/POM extracts as described by Ghosh (2014). A simple mass balance calculation is used to convert extract concentrations to concentrations within the PE/POM. Then the porewater concentration is estimated as the ratio of the PE/POM concentration and a literature derived polymer-water partition coefficient (K_{PS}). The PRCs were used to assess the extent of equilibrium achieved during the period that the samplers were in contact with the sediment. Some of the strongly hydrophobic compounds, in particular, the higher chlorinated PCB congeners, did not achieve equilibrium during the period of contact and a mathematical correction was performed to calculate the true equilibrium concentrations (Fernandez et al. 2009). Uncertainty in the porewater concentrations arises from the fact that the source of the K_{PS} values are regression relationships derived in other experiments with these samplers (Ghosh 2014; Choi et al. 2013; Fernandez 2012; Cornelissen et al. 2008). Another source of uncertainty for the dioxin/furan concentrations is that PRCs were not used in the POM samplers and therefore no equilibrium corrections were made. There is a possibility that the dioxin/furan concentrations are biased low; however, if non-equilibrium occurred, it is more of a concern for the higher chlorinated dioxin/furan congeners that have much lower toxicity equivalent factors compared to, for example, 2,3,7,8-TCDD. For PCBs, non-equilibrium occurred infrequently and only among the higher chlorinated congeners (i.e., hexa-, hepta-, octa-, nona-, and deca-chlorinated congeners).

The uncertainty of K_{PS} values used in this SQT assessment is further described in the USEPA's passive sampling user's manual (USEPA/SERDP/ESTCP 2017). Estimated values of K_{PS} are expected to be accurate within 0.3 log units (i.e., a factor of 2) even for highly hydrophobic compounds. The resulting error is similar to other environmental partition coefficients. Therefore, porewater concentrations could be as much as two times greater or lower than the calculated values presented here. As an uncertainty analysis, porewater concentrations for organic chemicals were doubled and then compared to the acute and chronic criteria and screening values used in this assessment (Appendix A; Table A-11). The chronic exceedance counts increased for PCBs and DDx beyond that shown in Table 6-7. For PCBs, doubling the concentrations resulted in five additional samples exceeding the chronic criteria for a total of eight out of 30. For total DDx, doubling the concentrations resulted in 15 additional samples exceeding the chronic criteria for a total of 29 out of 30. The doubling also resulted in one sample (i.e., station 154) exceeding the chronic freshwater SQuiRT value for 2,3,7,8-TCDD (Appendix A; Table A-11) since the estimated value changed of 9.35 pg/L exceeds 10 pg/L when doubled. No additional acute exceedances resulted from the doubling. The PAH SumTU values were also be doubled to evaluate the uncertainty. Doubling results in one additional sample (i.e., station 151) with a PAH SumTU that exceeds one.

Modeling of acute toxicity indicated a strong association between sediment and porewater chemistry and 10-day survival. Because there are only two stations that exhibited acute toxicity, there is uncertainty that the relationship is real or could have happened by chance. However, a review of the historical data shows a clear relationship between 10-day *A. abdita* survival and sediment chemistry in the NBSA (Appendix A) so it is very likely that the same relationship exists with *L. plumulosus* as both organisms have demonstrated a similar ability to identify toxic sediments (Schlekat et al. 1995). There is also uncertainty in the model fitting since all but two stations had very high 10-day survival. The PC1 scores of the acutely

toxic stations, 151 and 154, are 7.3 and 5.6 respectively. The next highest PC1 score for the non-acutely toxic samples is 3.52. There are four non-acutely toxic stations with PC1 scores that exceed 3 so it is clear that the threshold for acute toxicity is somewhere between 3.5 and 5.6. The sediment model predicts toxicity at PC1 scores exceeding 5 and identified 21 Phase III locations as potentially acutely toxic. If the true PC1 score threshold for toxicity were lower, for example, 4, then an additional 13 Phase III locations would also be classified as potentially acutely toxic for a total of 34, 13.4% of all samples. If the true PC1 score threshold for toxicity was 3.6, then an additional 16 Phase III locations would also be classified as potentially acutely toxic for a total of 37, 14.6% of Phase III samples.

There is uncertainty associated with the results of toxicity tests. Endpoint response to sediments is not necessarily indicative of contaminant effects (Kennedy et al. 2009). A reduction in endpoint values (i.e., survival, growth, number of young) compared to control could be caused by other factors including inherent random variability that could affect both the tested samples and/or the control samples. The 10-day acute toxicity tests have control acceptability criteria that require ≥ 90 % survival; therefore, control variability is tightly controlled in these tests. Nonetheless, clean reference samples have been shown to exhibit 10-day amphipod survival less than control (Kennedy et al. 2009).

The 28-day tests are subject to more variability. In an interlaboratory comparison, Kennedy et al. (2009) found greater inconsistency among chronic endpoints as compared to acute testing results. For 28-day survival, Kennedy et al. (2009) and Eickhoff et al. (2014) found fairly low variability among laboratory control samples; within replicate, CVs ranged from 2 to 11%. However, among test sediments, toxicity classifications differed among duplicate testing results (Kennedy et al. 2009). For example, a sediment sample from Newark Bay, tested three different times exhibited 28-day survival significantly less than control (i.e., toxic) in the first test and comparable to control in subsequent tests (i.e., non-toxic). A clean reference sample exhibited 28-day survival of 60%, which would result in classification as toxic.

The sublethal endpoints are subject to considerably more variability in both control and test sediments. There are no control acceptability requirements that control variability in these endpoints. Eickhoff et al. (2014) subjected five control sediments to 28-day tests with *L. plumulosus* and found growth and reproduction endpoints were more variable both among control samples and within batch replicates. Mean growth in control samples ranged from 0.7 to 1.8 milligrams per organism and the mean CV among replicates was 30%. Reproduction had a 10-fold range within control sediments (0.8 to 8.4 juveniles per surviving adult) and the mean CV among replicates was 79%. Kennedy et al. (2009) found significant variability in control sediment for growth (CV = 56%) and reproduction (CV = 63%), as well as significant variability in an interlaboratory comparison of the growth and reproduction endpoints. *L. plumulosus* reproduction is known to be highly dependent on grain size (McDonald et al. 2010). This variability leads to uncertainty in the interpretation of these chronic tests.

Variability in the sublethal endpoints was also demonstrated in split sample testing conducted by the USEPA on a subset of three SQT samples (USEPA 2017b). Growth was found to be significantly different than control in samples from stations 144, 156, and 157 (Table 6-4). However, the split results indicated growth greater than control for these same sediments. This inconsistency is not unlike what has been documented in the literature and confirms the inherent uncertainty in the interpretation of sublethal endpoints.

For the BIC comparisons to reference, there is uncertainty due to the difference in sampling depths, 2 cm for Jamaica Bay and 6 inches in the NBSA, which may cause differences in abundance and richness in

the samples. Both programs incorporated three grab samples per location for taxa enumeration. In both programs, BIC samples were sieved with a 0.5 mm sieve.

There is uncertainty regarding the incorporation of the 2013 REMAP data from Jamaica Bay into the reference dataset. As noted in Section 2.3.1, there was a significant decline in the BIC metrics when 2013 REMAP data are compared to 2008 REMAP data (and prior) in the NBSA. This same pattern was also observed in Jamaica Bay. This decline affected only BIC measures as there was no change in the incidence of toxicity. The mean chemical concentrations presented in Appendix A are similar between the two programs. It is not known why this apparent decline in BIC occurred, but given it was observed in both estuaries it is due to either of two possibilities: 1) a change in sampling methodology (unlikely, due to REMAP using similar methods since 1993); or 2) changes caused by a regional event. In fact, the 2013 samples were collected after a significant meteorological event had occurred in October 2012, Hurricane Sandy. It is possible the storm affected the BIC. In the NBSA, these effects, if real, were short-lived because the 2015 dataset indicated a full recovery. There are currently no available data to infer whether the BIC has rebounded in Jamaica Bay since 2013. To evaluate this uncertainty, the 2013 REMAP data were removed from the Jamaica Bay reference dataset, and the comparisons to the NBSA were reanalyzed (see Appendix A). It was found that removal of the 2013 data had no impact on the evaluation. There were no additional exceedances of the reference envelope nor changes in SQT scoring, and the hypothesis testing results were similar in that there was no indication the NBSA had lower mean BIC values than Jamaica Bay.

There is uncertainty regarding whether the 30 SQT stations are representative of the larger Phase III surface sediment data sets. This was evaluated in two ways, first, by comparing summary statistics of selected chemicals for the 30 SQT stations and the remaining Phase III samples. Mean concentrations were found to be similar for the SQT samples and the remaining samples. For example, the mean 2,3,7,8-TCDD concentration in SQT samples is 91.3 ng/kg while for the remaining Phase III samples, it is 86.6 ng/kg. Other chemical concentrations compare similarly. The mean SQT versus remaining samples are as follows respectively, total PCB congeners (588 micrograms per kilogram [µg/kg~~,~~], 465 µg/kg), total 2,4' and 4,4' DDx (98.7 µg/kg, 135 µg/kg), total PAHs (13,000 µg/kg, 12,000 µg/kg), mercury (1.92 mg/kg, 2.4 mg/kg). The mean and median concentrations for other chemicals are also similar as well~~ as total organic carbon and percent fines.~~. The mean percent fines for the SQT samples (60.3%) is higher than for the remaining Phase III samples (55.5%) and mean TOC for the SQT samples (40,100 mg/kg) is also higher than for the remaining Phase III samples (21,400 mg/kg). This could be due in part to the finer grained samples in the SQT data set. In addition, an extreme outlier for TOC (151,000 mg/kg from station 154) occurs in the SQT data. With the outlier excluded, the mean TOC of the SQT samples is 36,300 mg/kg. Higher TOC may decrease the bioavailability of organic compounds. The data sets were also evaluated in the principal component analysis and were found to have a similar range and chemical composition as the remaining sediment samples. These evaluations indicate that the SQT samples are representative of the full Phase III data set and the NBSA surface sediments. The details of these evaluations are found in Appendix A. Because the SQT locations are representative of the overall Phase III dataset, conclusions based on the integrated SQT lines of evidence (sediment chemistry, toxicity and benthic community assemblage) can be applied within the NBSA for remedial decision-making at spatial scales smaller than the assessment zones evaluated in this BERA.

6.4.2 Invertebrate Tissue Risk Uncertainties

Sources of uncertainty that influenced invertebrate tissue risk characterization included uncertainties in the analytical results, data evaluation, effects assessment, and interpretation of the risk estimates. Many of these sources of uncertainty are generic in nature and inherent in the risk assessment process, particularly with relying on the use of literature-based TRVs and the computation of HQs to make conclusions on population-level risks.

Inherent practices in risk assessment that will bias high the estimates of exposure to be reasonably conservative include:

- ~~The use of an upper exposure value (i.e., 95% UCL) to calculate dietary exposure concentrations~~
- The use of a maximum values in the place of a UCL when limited data are available
- The assumption that blue crab populations feed exclusively or get all of their seasonal and lifetime exposure to COPECs from the NBSA.

Uncertainties associated with the invertebrate tissue TRVs are discussed in Appendix D. In general, there are limited whole-body tissue TRVs available for invertebrate organisms, making it difficult to ~~find~~derive TRVs that are specific and relevant for particular groups such as bivalves and crustaceans. This renders most invertebrate TRVs uncertain. The goal of using different sets of TRVs is to try and bound those uncertainties. However, application of a TRV for one group to another ~~that have~~with very different life histories, exposure scenarios, and potential toxicological/adaptive sensitivities to COPECs can make the resulting risk estimates highly uncertain.

~~Of particular note is the high degree of uncertainty associated with the use of oyster (i.e., bivalve) TRVs in the LPR FFS to assess potential risks to blue crab for 2,3,7,8-TCDD (derived from Cooper and Wintermyer 2009) and PCBs (derived from Chu et al. 2000; Chu et al. 2003).~~For 2,3,7,8-TCDD, the main uncertainties with the crayfish TRVs from Ashley et al. (1996) and used in the LPRSA BERA (Windward 2019) are: mortality is generally not the most sensitive endpoint based on chemical exposure; tissue concentrations were not reported; the test individuals were wild-caught; only three or four individuals were exposed per treatment; and application of the TRVs to non-crustaceans is highly uncertain. The TRVs for 2,3,7,8-TCDD used in the LPR FFS and derived from a NBSA site-specific study of Eastern oyster reproduction (Wintermyer and Cooper 2003) also have uncertainties that include: reported tissue concentrations were based on one composite of seven oysters from each site; the endpoint; fertilized egg development was based on one sample; the presence of other contaminants were not considered or evaluated; a true control group was not used for comparison; and application of the TRVs to non-bivalves is highly uncertain. The species evaluated for PCB invertebrate toxicity were more diverse given that an SSD based on 10 species was used to derive the TRVs in the LPRSA BERA (Windward 2019) and this reduces inter-species toxicity uncertainty. However, as with 2,3,7,8-TCDD, Eastern oyster reproductive toxicity was the basis of the alternate PCB TRVs (Chu et al. [2000, 2003] as cited in the LPR FFS). This is a highly conservative and uncertain TRV application, given the very different life histories and exposure potential between sessile filter-feeding oysters and mobile blue crab. The use of ~~this TRV~~ both the 2,3,7,8-TCDD and the total PCB TRVs derived from studies on Eastern oysters produces substantially different risk estimates (i.e., HQs) for blue crab than does the application of the ~~NBSA TRVs for these COPECs in blue crab, where a~~ crustacean ~~TRV was used.~~

~~While~~survival TRV for 2,3,7,8-TCDD, and the 10 species SSD for survival, growth, and reproduction for PCBs. However, given that TRVs do not exist specifically for blue crab and are limited for ecologically relevant endpoints other than survival ~~may be appropriate for blue crab, it is clear that blue crab are not as sensitive to dioxin-like effects as the eastern oyster tested by Cooper and Wintermyer (2009). The eastern oysters are not known to presently (or~~in crustaceans in ~~the recent past) occur in the NBSA. This is likely due to a combination of habitat alterations, pathogens (i.e., Dermo and MSX that wiped out much~~general, the use of this range of ~~the oyster populations in mid-Atlantic estuaries in the 20th century), and impacts from multiple contaminants. On the other hand, blue crab have been and continue to be abundant in Newark Bay and its tributaries despite the varying levels of contamination and other stressors. As such, it is not reasonable to assume they are as sensitive as the eastern oyster to dioxin-like compounds or other contaminants, or substantial impacts to the populations would be expected~~TRVs was used for risk bounding.

Whole-body tissue concentrations of total metals (other than organometals, including methylmercury, organo-selenium, and butyltins) are poorly predictive of adverse effects for several reasons: toxicity is caused by specific metal species, invertebrates store excess metals in non-toxic compartments, and toxicity is strongly dependent on the exposure rate and ~~exposure~~ pathway. The use of a tissue residue approach for metals (except for methylmercury and selenium) is highly uncertain because of the wide range of strategies used by organisms to store, detoxify, and excrete bioaccumulated metals (e.g., fish and invertebrates may regulate their body burdens of some metals; although, metals regulation, and the strategy thereof, is species and metal specific) (Adams et al. 2011; USEPA 2007b). For these reasons, USEPA risk assessment guidance for metals and recent expert guidance conclude that comparison of whole-body metals tissue concentrations to literature-reported whole-body effects thresholds is not sufficiently robust for drawing risk conclusions (Adams et al. 2011; USEPA 2007b). Further details on key uncertainties for individual metals are presented in Appendix D.

With respect to the treatment of non-detects for invertebrate EPCs, the concentrations of sum components (e.g., PCB congeners) that were not detected were assumed to be zero when calculating totals. The effect on HQs of using one-half the ~~detection~~quantitation limit or the full ~~detection~~quantitation limit (ND=1) was evaluated and there were no changes to NBSA-wide or zone LOAEL tissue HQs based on either NBSA or LPR FFS-based TRVs for softshell clam or blue crab. Summary tables of HQ results from this sensitivity analysis on the treatment of non-detects are ~~found~~presented in Appendix F.

6.5 Weight of Evidence Risk Findings for Benthic Invertebrates

For benthic invertebrate infaunal communities, the risk characterization outcome is based on integrating the WOE from the SQT assessment for infaunal communities. Additional LOE for benthic (infaunal) communities include only tissue HQs for polychaete worms, which were considered a secondary LOE as discussed in Section 6.2.3. For epibenthic clams (bivalves) and blue crab, tissue-based HQs were the primary LOE available and were used to ~~draw~~evaluate COPECs with LOAEL HQs ≥ 1 (preliminary COCs; Table 6-21) and potential for unacceptable risk ~~conclusions for~~ to these receptors (i.e., shellfish).

Table 6-21. Summary of Preliminary COCs for Benthic Invertebrate Tissue Assessment

Preliminary COCs	Exposure Area	Range of LOAEL HQs					
		Invertebrate Tissue LOE					
		NBSA TRVs			LPR FFS TRVs		
		Polychaete Worm	Blue Crab	Softshell Clam	Polychaete Worm	Blue Crab	Softshell Clam
2,3,7,8-TCDD	NBSA-wide	<1	<1	<1	**9**	**19**	**5**
	Zones	--	<1	<1	NC	**13 - 27**	**4 - 7**
Total PCBs	NBSA-wide	<1	<1	<1	**4**	**21**	**8**
	Zones	--	<1	<1	NC	**17 - 28**	**5 - 17**
Total DDx (2,4 & 4,4)	NBSA-wide	<1	**1**	<1	<1	<1	<1
	Zones	--	<1- **1**	<1	NC	<1 - **1**	<1
Total DDx (2,4 & 4,4) Alternative TRV	NBSA-wide	<1	**12**	**3**	NC	NC	NC
	Zones	--	**10 - 16**	**1 - 5**	NC	NC	NC
Dieldrin	NBSA-wide	<1	<1	<1	<1	<1	<1
	Zones	--	<1	<1	NC	<1 - **1**	<1
Total HMW PAH	NBSA-wide	<1	<1	**2**	<1	<1	**2**
	Zones	--	<1	<1	NC	<1	<1
Arsenic	NBSA-wide	**4**	**4**	**9**	NC	NC	NC
	Zones	--	**3 - 5**	**4 - 14**	NC	NC	NC
Cadmium	NBSA-wide	<1	**1**	<1	NC	NC	NC
	Zones	--	**1 - 2**	<1 - **1**	NC	NC	NC
Chromium	NBSA-wide	<1	<1	**1**	NC	NC	NC
	Zones	--	<1	**1 - 2**	NC	NC	NC
Copper	NBSA-wide	NC	NC	NC	<1	**3**	**1**
	Zones	NC	NC	NC	NC	**3**	<1 - **2**
Lead	NBSA-wide	<1	<1	<1	<1	<1	**11**
	Zones	--	<1	<1 - **2**	NC	<1	**4 - 36**
Mercury	NBSA-wide	<1	**1**	**2**	<1	**1**	**2**
	Zones	--	**1 - 2**	**1 - 3**	NC	**1 - 2**	**1 - 3**
Methylmercury	NBSA-wide	<1	**1**	<1	<1	**1**	<1
	Zones	--	**1 - 2**	<1	NC	**1 - 2**	<1
Nickel	NBSA-wide	<1	<1	**3**	NC	NC	NC
	Zones	--	<1	**2 - 5**	NC	NC	NC
Selenium	NBSA-wide	<1	**2**	**2**	NC	NC	NC
	Zones	--	**2**	**2**	NC	NC	NC
Silver	NBSA-wide	<1	**1**	<1	NC	NC	NC
	Zones	--	**1 - 2**	<1 - **1**	NC	NC	NC
Zinc	NBSA-wide	<1	<1	<1	NC	NC	NC
	Zones	--	<1	<1 - **3**	NC	NC	NC

Notes:
NC = not calculated; TRV not available; -- = COC not evaluated

Additional LOE include site-specific measurements of reduced reproductive success in oysters in Newark Bay (Wintermyer and Cooper 2003~~).~~), and consideration of background sediment concentrations (see Section 4.4.2). A summary of the risk characterization and LOE for benthic invertebrates is presented in Table 6-~~21~~22. For benthic invertebrate (infaunal) communities, the results of the SQT integrate all available LOEs and were used to make risk conclusions. The overall SQT scores (described in Section 6.1.1) and summary of SQT findings (presented in Table 6-17) indicate impairment in localized areas of the NBSA. The individual LOEs contributing to the increased SQT scores and conclusion of localized risk are primarily sediment toxicity test results and exposure-response models from the SQT, which appear to

be a function of localized elevated concentrations of multiple co-occurring COPECs. These include dioxins/furans, PCBs, pesticides (particularly total DDx), PAHs, and many inorganics/metals (particularly, but not limited to, arsenic, cadmium, chromium, and mercury). The areas of localized risk (see Figures 6-6 and 6-8) represent a small proportion of the NBSA. Exposure of individual organisms to these areas is considered unlikely to substantially impact the overall benthic invertebrate communities and populations within the NBSA.

For shellfish (softshell clam and blue crab), the risk conclusions are based on the tissue HQ LOE (Table 6-~~21~~22). Additional LOE for clams include site-specific toxicity testing as reported in Wintermyer and Cooper (2003). ~~While a low number of~~ LOAEL HQs were >1 for 2,3,7,8-TCDD, total PCBs, total DDx, dieldrin, and various metals for blue crab, and for 2,3,7,8-TCDD, total PCBs, total DDx, total HMW PAHs, and various metals for softshell clam~~, the~~. The tissue-based risks ~~to epibenthic invertebrates~~, based on the relative magnitude of the LOAEL HQs, are likely limited to 2,3,7,8-TCDD and PCBs in bivalves (represented by softshell clam~~). The increased risk for shellfish populations~~), and 2,3,7,8-TCDD, total PCBs, and total DDx in blue crab, and possibly a few metals in localized areas, particularly arsenic, mercury/methylmercury, and lead.

For bivalves, the unacceptable risks are ~~listed as possible~~likely for 2,3,7,8-TCDD and total PCBs due to tissue HQs greater than 10 calculated using ~~both NBSA and~~ LPR FFS TRVs~~. The~~ and site-specific evidence of potential reproductive impairment in oysters (Wintermyer and Cooper 2003). For 2,3,7,8-TCDD, the LPR FFS TRVs are based ~~on a~~the Wintermyer and Cooper (2003) study ~~by Wintermyer and Cooper (2003)~~ that evaluated the bioaccumulation of dioxins/furans and PCBs and concurrent reproductive success of transplanted oysters in Newark Bay and other study sites in the NY/NJ Harbor Estuary. The study found diminished reproductive success in oysters deployed in the Bay. While it was focused on dioxins/furans and PCBs, the authors of this study did not take into account the various other contaminants present in the study system. ~~However, a~~A follow-up laboratory study—Cooper and Wintermyer (2009)—was used to associate similar reproductive effects observed in the prior field study with tissue data for dioxin/furan/PCB TEQ in oysters. This study ~~is~~reported effect levels similar to, but about two times greater, than the ~~source of the NBSA TRVs used in this BERA for bivalves.~~LOAELs from Wintermyer and Cooper (2003). It is unclear whether ~~these~~the reported effects translate to impairments to the maintenance of bivalve populations in the NBSA, but there is some potential for increased risk of adverse effects to individual bivalve reproduction. ~~For blue crabs, unacceptable risk is considered unlikely based on the low magnitude of the tissue-based HQs. For 2,3,7,8-TCDD and PCBs, blue crab tissue-based HQs calculated using LPR FFS TRVs were considered a secondary LOE due to the uncertainties associated with applying these bivalve TRVs to crustaceans (see Section 6.3.2).~~

~~7 FISH ASSESSMENT~~

The NBSA TRVs for 2,3,7,8-TCDD are based on crayfish data (Ashley et al. 1996), which is more relevant to blue crab (crustaceans) than bivalves.

For several metals, LOAEL HQ > 1 were also predicted for bivalves. However, the magnitude of the HQs was generally low except for arsenic and lead, and there is significant uncertainty in the tissue HQ approach for metals. Unacceptable risk was considered possible for mercury, given that LOAEL HQs > 1 were predicted for all scenarios using both NBSA and FFS TRVs, and there is a lower degree of uncertainty in the tissue TRVs for mercury and methylmercury. Some of these metals (i.e., selenium and silver) are present at concentrations near or below background levels in sediment. Unacceptable risk is considered possible in some cases, consistent with the SQT evaluation.

For blue crabs, unacceptable risk is considered likely for 2,3,7,8-TCDD and PCBs, based on substantial exceedance of the LPR FFS TRVs. For total DDx, LOAEL HQs close to or greater than 10 were predicted using the alternate NBSA TRV; however, LOAEL HQs were never greater than 1 using NBSA or LPR FFS TRVs. Therefore, total DDx risks to blue crab are considered unlikely. LOAEL HQs > 1 were also predicted for several metals as well. Similar to the evaluation for softshell clam, the magnitude of the metals HQs was generally low and there is significant uncertainty in the tissue HQ approach for metals. Unacceptable risk was considered possible for blue crab exposed to mercury and methylmercury, based on the same rationale as for softshell clam.

~~8~~7 FISH ASSESSMENT

The fish community of the NBSA includes species at various trophic levels of the food web. They consume invertebrates and other lower-trophic level organisms but, in turn, are fed upon by upper-trophic level birds and mammals. Similar to invertebrates, there are various taxonomic feeding guilds for fish including benthic-feeding (i.e., benthivores), omnivorous forage fish, and pelagic predatory fish. Fish also form part of the food web for other fish and upper-trophic level organisms such a birds and mammals. This section includes the risk assessment for fish in the NBSA.

The following are the Assessment Endpoints and Risk Questions for fish that are addressed in this section.

Assessment Endpoint: Survival, growth, and/or reproduction of fish.

Risk Question 1: *Are fish communities in the NBSA different from those found in similar nearby water bodies and comparable to what's expected in mid-Atlantic estuaries?*

Risk Question 2: *Are COPEC concentrations in fish tissues from the NBSA greater than toxicity benchmarks for the survival, growth, and/or reproduction of fish?*

Risk Question 3: *Are COPEC concentrations in porewater, surface water, and sediment from the NBSA greater than benchmarks for the survival, growth, and/or reproduction of fish?*

Risk Question 4: *Does the daily dose of COPECs received by fish (including forage fish, benthic omnivore, pelagic invertivore, and pelagic piscivore) from consumption of the tissues of prey species and from other media in the NBSA exceed the TRVs for survival, growth, and/or reproduction of fish? If yes, what are the probabilities of effects of differing magnitude for survival, growth, and/or reproduction of fish?*

Risk Question 1 is addressed by a qualitative analysis of available adult and juvenile fish and ichthyoplankton community data from a study conducted in the NY/NJ Harbor Estuary including site-specific data collected from the NBSA as presented in Section 7.1. Risk Question 2 is addressed by the fish tissue-based assessment presented in Section 7.2 and the fish egg tissue assessment presented in Section 7.4. Risk Question 3 is addressed in the surface water assessment in Section 5. Risk Question 4 is addressed in the fish dietary risk assessment in Section 7.3. The life history profiles for the fish receptors identified for the tissue and dietary risk assessments are provided in Appendix B.

A summary of the fish pathology investigation that was conducted as part of the BERA fish sampling program is provided in Section 7.5 as a LOE for the fish assessment, particularly to help assess the overall condition of fish populations in the NBSA. The full fish pathology reports are provided as Attachment 5 to this BERA. An uncertainties assessment is provided for the overall fish assessment in Section 7.6, and the WOE summary of findings for fish is provided in Section 7.7. Additionally, a field investigation on potential contaminant effects on mummichog reproduction (Bugel et al. 2010) is included for perspective for the WOE in Section 7.7.

The species for which data were collected under the RI/BERA and evaluated in this assessment were identified as predominant resident and migratory fish that inhabit or utilize the NBSA, and as important representatives of Bay fishery. They were selected based on the substantial historical data and

information available on the Bay fishery (Section 2.8), including the various life histories of fish from ichthyoplankton to adult stages, and consideration of the EFH designations by the National Marine Fisheries Service for the NBSA. These fish include species from different feeding guilds, and with different levels of potential exposure to COPECs in sediments and the food web of the system. For these reasons, the assessment provided in this section is fully representative of the potential risks to NBSA fish populations.

~~8.1~~7.1 Fish Community Assessment

~~8.1.1~~7.1.1 Available Data

The USACE-New York District conducted an extensive Aquatic Biological Survey (ABS) of the NY/NJ Harbor Estuary to assess the seasonal distribution and abundance of the fish community as well as habitat information (salinity, water temperature, oxygen levels) as a basis for informed dredging management practices and to promote the protection of fish assemblages utilizing existing habitats of the Harbor and near shore habitats of the New York Bight.

The ABS was a long-term, diverse and spatially robust sampling program that included bottom trawl surveys of demersal or near bottom occurring finfish (~~2002~~2001 – 2010 dataset) and ichthyoplankton tows using a benthic sled equipped with a plankton net to collect fish eggs and larvae near the bottom (~~2002~~2001 – 2011 dataset; USACE 2015b). Sampling locations were divided into three subregions: 1) Newark Bay and Arthur Kill; 2) the Upper New York Bay (South Brooklyn, Port Jersey, Kill Van Kull); and 3) the Lower New York Bay (Ambrose Channel, Chapel Hill South Channel, Raritan Bay East Reach, East Bank, Swash Channel Range, Old Orchard Shoals).

The data from these surveys were used to compare and contrast the fish assemblages at different life stages (i.e., adult/juvenile and ichthyoplankton), including seasonal occurrence, diversity, and abundance of species between the survey locations. In addition, the USACE evaluated the fish community found in their studies relative to those typical of mid-Atlantic estuaries. This dataset provides a unique opportunity to assess the relative occurrence, diversity, and abundance of fish species in the NBSA region, relative to other regions in the NY/NJ Harbor Estuary and in general for fisheries in mid-Atlantic estuaries.

~~8.1.2~~7.1.2 Qualitative Evaluation of Fish Community Data

~~8.1.2.1~~7.1.2.1 Water Quality and Habitat Information

Water quality data from the ABS show that salinities were consistently lowest in the Arthur Kill/Newark Bay and highest in the Lower Bay throughout the seasons. ~~DO~~Dissolved oxygen trends were similar across the three Harbor regions; gradually decreasing from January through April, and then rapidly decreasing from April through June as water temperatures increased.

The bottom water temperature was slightly lower in the Arthur Kill/Newark Bay from December through February and slightly higher from March through June than in the other regions of the Harbor. The warmest water temperatures occur in the Arthur Kill/Newark Bay region during the spring and into early summer (April to July). The Arthur Kill/Newark Bay region also had some of the coolest bottom temperatures in January. The Arthur Kill/Newark Bay and, to a lesser extent, the Upper Bay salinities and

temperatures are influenced by freshwater runoff from the Raritan, Passaic, Hackensack, and Hudson Rivers.

The bottom substrate in the Arthur Kill/Newark Bay region is comprised of fine grain silt and clay, while the Lower Bay is primarily composed of sand and coarse-grained sands. The Upper Bay consists of a mixture of finer clay and silt, and sand sediments. The Arthur Kill/Newark Bay and Upper Bay regions have a larger percentage of maintained deep channels and berthing areas compared to the Lower Bay, which is dominated by shallows/shoals with a smaller percentage of maintained channel areas.

~~8.1.2.2~~7.1.2.2 Adult and Juvenile Fish Community

~~8.1.2.2.1~~7.1.2.2.1 Abundance

The ABS dataset shows that many of the numerically dominant fish species collected during the winter, such as white perch, striped bass, and winter flounder, were concentrated in Arthur Kill/Newark Bay (Table 7-1; USACE 2015b). Therefore, overall abundances of fish of all species in the winter in Arthur Kill/Newark Bay are nearly twice as high as that collected in Lower Bay and nearly four times as high as winter fish abundances in Upper Bay. Total winter fish catch per unit effort over the nine years of sampling averaged 96 fish/10 minute trawl in Arthur Kill/Newark Bay, 58 fish/10 minute trawl in Lower Bay, and 28 fish/10 minute trawl in Upper Bay.

Table 7-1. Relative Abundance of Adult and Juvenile Fish in the NY/NJ Harbor Estuary Subregions: December-March (Winter) 2001-2011

Species*	Average Catch-Per-Unit-Effort (No. Fish/10 Minute Trawl)		
	Arthur Kill/Newark Bay	Lower Bay	Upper Bay
	Adult/juveniles	Adult/juveniles	Adult/juveniles
White perch	80	0.01	0.8
Striped bass	19	0.1	5.1
Winter flounder	4.1	1.6	3.0
Red hake	2.1	1.2	2.9
Alewife	1.5	16.2	3.7
Spotted hake	1.5	5.2	3.6
Blueback herring	1.5	7.2	4.2
Atlantic silverside	0.7	6.5	0.6
Bay anchovy	0.4	0.8	0.5
Windowpane	0.3	0.5	1.0
American shad	0.3	0.5	1.2
Silver hake	0.1	1.6	0.2
Atlantic menhaden	0.1	1.5	0.2
Smallmouth flounder	0.1	1.2	0.3
American sandlace	0	11	0.1
Grubby	0	0	0.2
Atlantic croaker	0	0	0.6
Atlantic herring	0	0.8	0.4
Spotted hake	0	0	0.5

Notes: *Species that collectively comprised 95% of all fish collected in a subarea; From USACE (2015)

In spring, fish densities were highest in Upper Bay, with a mean total of 115 fish/10 minute trawl compared to 92 fish/10 minute trawl in Arthur Kill/Newark Bay and 64 fish/10 minute trawl in Lower Bay. Bay anchovy was dominant in all three NY Harbor locations in spring, and spotted hake as well as Atlantic herring were also common (Table 7-2; USACE 2015b).

Table 7-2. Relative Abundance of Adult and Juvenile Fish in the NY/NJ Harbor Estuary Subregions: April-July (Spring) 2001-2011

Species*	Average Catch-Per-Unit-Effort (No. Fish/10 Minute Trawl)		
	Arthur Kill/Newark Bay	Lower Bay	Upper Bay
	Adult/juveniles	Adult/juveniles	Adult/juveniles
Bay anchovy	29	44	80
Spotted hake	23	6.8	18
Atlantic herring	23	2.7	14
Striped bass	4.3	0.1	2.8
Alewife	3.8	2.6	1.7
Winter flounder	2.5	2.0	3.1
White perch	2.3	0	0.3
Blueback herring	2.3	0.9	2.7
Red hake	2.1	0.5	2.8
Atlantic tomcod	1.1	0.02	5.5
Windowpane	0.6	0.6	1.4
Scup	0.01	1.2	1.2
Atlantic butterfish	0	0.9	0
Striped searobin	0	0.7	0.0
American sandlance	0	0.1	1.5

Notes: *Species that collectively comprised 95% of all fish collected in a subarea; From USACE (2015)

~~8.1.2.2.2~~7.1.2.2.2 *Diversity*

Generally, alewife, blueback herring, spotted hake, winter flounder, and red hake occur in all Harbor areas during the winter (Table 7-1; USACE 2015b). Winter species communities were similar in the Arthur Kill/Newark Bay regions due to large abundances of white perch and striped bass, species that were much less common in the Upper Bay and Lower Bay (Table 7-1; USACE 2015b). White perch are uncommon in the Upper and Lower Bay, which have more saline waters compared to Arthur Kill/Newark Bay. Although the Upper and Lower Bay fish densities were lower than that of Arthur Kill and Newark Bay, the communities were more diverse.

In spring, Harbor fish communities were composed of bay anchovy, spotted hake, Atlantic herring, alewife, blueback herring, and winter flounder (Table 7-2; USACE 2015b). Spotted hake accounted for more than half the fish in Arthur Kill and bay anchovy and Atlantic herring comprising more than half the fish in Newark Bay. The spring demersal fish communities in Upper Bay and Lower Bay were dominated by bay anchovies, spotted hake and Atlantic herring. Spring species diversity was similar across the three Harbor regions.

~~8.1.2.3~~7.1.2.3 Ichthyoplankton (Egg/Larvae) Community

The ABS found that the numerically dominant egg and larvae species collected included gobies, wrasses, Atlantic menhaden, searobins, weakfish, and windowpane (Tables 7-3 and 7-4; USACE 2015b).

Table 7-3. Relative Abundance of Eggs and Larval Fish in the NY/NJ Harbor Estuary Subregions: December-March (Winter) 2001-2011

Species*	Average Catch-Per-Unit-Effort (No. Eggs/Larvae Per 1,000m^3)		
	Arthur Kill/Newark Bay	Lower Bay	Upper Bay
	Egg/larvae	Egg/larvae	Egg/larvae
Grubby	34	48	41
Winter flounder	17	75	61
Atlantic tomcod	13	1.9	4.9
American sandlance	10	86	32
Atlantic croaker	8.9	2.2	13
Fourbeard rockling	6.3	4.1	12
Rock gunnel	6.2	11	9.1
Atlantic herring	4.8	2.7	3.7
Summer flounder	4.5	4.2	5.6
Atlantic menhaden	3.3	5.2	5.5
Gadidae (cod)	2.9	4.1	10
Gobiidae (goby)	0	36	3.4

Notes:
*Species that collectively comprised 95% of all fish collected in a subarea
Eggs/larvae category includes eggs, yolk sac, post-yolk sac
From USACE (2015)

Table 7-4. Relative Abundance of Eggs and Larval Fish in the NY/NJ Harbor Estuary Subregions: April-July (Spring) 2001-2011

Species*	Average Catch-Per-Unit-Effort (No. Eggs/Larvae Per 1,000m^3)		
	Arthur Kill/Newark Bay	Lower Bay	Upper Bay
	Egg/larvae	Egg/larvae	Egg/larvae
Gobiidae (goby)	659	451	393
Labridae (wrasse)	566	1,146	1,053
Atlantic menhaden	238	1,135	592
Weakfish	171	514	124
Windowpane	143	471	266
Searobin species	118	1,022	291
Winter flounder	69	376	120
Tautog	35	245	85
Hogchoker	25	436	51
Clupeidae (shad/herring)	14	69	244
Gadidae (cod)	10	91	87

Notes:
*Species that collectively comprised 95% of all fish collected in a subarea
Eggs/larvae category includes eggs, yolk sac, post-yolk sac
From USACE (2015)

There were high larval abundances in Arthur Kill/Newark Bay non-channel locations of bay anchovy, gobies, and northern pipefish and lower larval abundances of winter flounder, windowpane, and grubby (USACE 2015b). The abundance and distribution of Atlantic menhaden, bay anchovy, windowpane, and wrasse eggs suggest spawning throughout the Harbor; while the abundance and distribution of fourbeard rockling, hogchoker, and weakfish eggs suggests those species spawn primarily in the Upper Bay and Newark Bay. Bay anchovy, goby, and northern pipefish larvae were relatively more abundant in Arthur Kill/Newark Bay than in Lower Bay and Upper Bay channel locations, whereas winter flounder, windowpane, and grubby larvae were not as abundant in Arthur Kill/Newark Bay (USACE 2015b). These spatial distribution patterns are potentially explained by physical tolerances to different salinity and water temperatures, and larval periods.

Winter flounder egg densities were lowest in the Arthur Kill/Newark Bay region and egg densities of fish species like searobins, cods, weakfish, Atlantic menhaden, and fourbeard rockling were lower at Arthur Kill/Newark Bay than other Harbor areas (USACE 2015b). Benthic habitats, sediment types, and disturbance histories differ among Harbor areas. Salinities also are lower in Arthur Kill/Newark Bay, therefore, site selection for spawning may include an aversion to disturbed benthic habitat and preference for higher salinities for egg and larval development (USACE 2015b)

A study of ichthyoplankton community in 1993 and 1994 in the lower Hudson River estuary, south of the George Washington Bridge and Battery Park, found that Atlantic tomcod and winter flounder young of the year dominated the fish community in early summer (May-July) while large numbers of young of the year striped bass were collected as part of a late summer assemblage (August-September; Able et al 1998). Black sea bass, American eel, cunner, and northern pipefish were also common.

Juvenile fish assemblages from a study in 1995 and 1996 in the Kill van Kull, Arthur Kill, and lower Hudson River found that juvenile silver perch, naked goby, mummichog, bay anchovy, Atlantic silverside, and winter flounder were commonly collected in benthic traps at Arthur Kill. Naked goby, Atlantic silverside, and winter flounder were also common in benthic traps at Kill van Kull and the lower Hudson River. The number of fish trapped was greatest at Arthur Kill compared to Kill van Kull and the lower Hudson River (Duffy-Anderson et al. 2003).

The USACE (2015a, 2015b, 2011) compared the fish assemblages found in their study to expected assemblages in terms of species occurrence and distributions of ichthyoplankton, juveniles, and adults, and concluded that they were comparable to those from other mid-Atlantic estuaries. These comparisons were made based on multiple surveys and studies conducted by Dr. Ken Able of Rutgers University and his associates (i.e., Able et al. 2011; Able and Fahey 2010; Able et al. 1998).

~~8.1.3~~7.1.3 Risk Characterization

The ABS dataset includes a wide variety of species representative of fish communities in the NBSA, including fish receptors evaluated in this BERA (winter and summer flounder, American eel, and white perch). Adult fish abundance and diversity in the Arthur Kill/Newark Bay were similar or greater than other locations in the NY/NJ Harbor. Juveniles, larvae, and eggs were commonly collected in the NBSA, indicative of ongoing reproduction within fish communities. Differences in species communities among the regions may be in part explained by physical differences such as bottom substrate, water temperature, salinities, and effects of freshwater inputs that can influence the presence of certain species.

~~8.2~~7.2 Fish Tissue Assessment

The fish tissue assessment was conducted for the following fish species: combined mummichog/striped killifish (i.e., *Fundulus* ~~spp~~sp.), summer and winter flounder, American eel, and white perch. A limited number of Atlantic Menhaden and silverside samples were also collected in the fish sampling efforts for this BERA. However, these are seasonal forage fish that move throughout the Bay and were not target species for the fish risk assessment. They were primarily collected to support the bird and mammal food web model (i.e., dietary) assessments. ~~A sensitivity analysis is presented in the uncertainty assessment for the fish tissue and dietary assessments to contrast potential risks to these species relative to the outcome for the target fish species.~~

The COPECs evaluated for the fish tissue assessment are listed in Table 3-5 and include: 2,3,7,8-TCDD, total dioxin/furan TEQ, total PCBs (209 congeners), total PCB TEQ, total dioxin/furan/PCB TEQ, pesticides (total chlordane, total DDx, dieldrin, and hexachlorobenzene), and inorganics (arsenic, chromium, copper, lead, mercury/methylmercury, selenium, silver, and zinc). Cadmium, nickel, and PAHs, which are COPECs for fish and other receptors in this BERA, were only evaluated in the dietary assessment for fish (Section 7.3), as they were not detected substantially in fish tissue samples. As such, a tissue assessment was not appropriate.

Tissue chemistry EPCs for the fish species were compared to whole-body tissue TRVs to calculate HQs. A separate focused assessment was conducted to evaluate the results of white perch liver samples, and to contrast the assessment of those findings with those from the whole-body HQ assessment. This section summarizes the COPECs, describes the derivation of tissue exposure and effects concentrations, and presents the HQs for fish tissue.

~~8.2.1~~7.2.1 Exposure

EPCs were calculated for COPECs identified for each fish species evaluated. EPCs were calculated as UCLs using all available whole-body tissue data (i.e., individual and composite; calculated whole-body samples) for each fish species. UCLs were calculated using USEPA's ProUCL® statistical package (Version 5.1) (USEPA 2015b) as described in Section 4.2. If a dataset contained fewer than six detected concentrations, a UCL was not calculated; instead, the maximum concentration was used as the EPC. Table 4-11 presents the sample locations in each assessment zone for fish tissue. The summary statistics for fish tissue by species are presented in ~~Table~~Tables 4-~~12.~~7d-k The EPCs for fish tissue COPECs are presented in Appendix C.

~~8.2.2~~7.2.2 Effects

Fish tissue TRVs for the COPECs identified in Table 3-5 are provided in Table 7-5 and discussed in detail in Appendix D. Fish tissue TRVs were primarily derived from the LPRSA BERA TRV database (Windward 2019; Appendix E~~). The lowest acceptable LOAEL and highest NOAEL below the LOAEL from the same study based on marine organisms were chosen. If marine TRVs were not available, then the freshwater-based TRV selected by the LPRSA BERA was used.~~) and are consistent with the Final LPRSA BERA TRVs for COPECs common to both areas (NBSA and LPRSA). For the LPRSA BERA, Windward (2019) conducted a literature search for relevant toxicological studies, and derived acceptable NOAELs and LOAELs. Total chlordane and hexachlorobenzene were not identified as COPECs for the LPRSA BERA

and, as such, TRVs were not developed by the CPG. For the NBSA BERA, a total chlordane TRV based on Parrish et al. (1976) and a hexachlorobenzene TRV based on Schuytema et al. (1990) were identified, as described in Appendix D.

In addition, the CPG did not assess fish tissue risks for some metals that are considered essential to fish metabolism and are physiologically regulated in terms of concentrations in the body of fish. They also did not assess fish tissue risks from PAHs due to the rapid metabolism of these compounds and excretion of their breakdown products by fish following uptake. For these metals and PAHs, the CPG relied primarily on a fish dietary assessment, similar to that presented in Section 7.3 of this BERA.

In addition to the above selected TRVs for the NBSA BERA, TRVs derived by the USEPA for the LPR FFS (USEPA 2014) were also assessed. The LPR FFS did not evaluate fish tissue-based risks for arsenic, chromium, selenium, silver, zinc, total chlordane, or hexachlorobenzene.

The primary NBSA 2,3,7,8-TCDD/TEQ TRV for fish tissue was a SSD-derived 5th percentile value based on survival, growth, and reproduction studies in seven species (NOAEL = 0.000012 mg/kg and LOAEL = 0.00012 mg/kg). This SSD-derived value as based on the geomean 5th percentile value for several different distributions (Windward 2019). An additional alternative fish tissue 2,3,7,8-TCDD/TEQ TRV was also developed by Windward (2019) in the LPRSA BERA using the same SSD dataset, but the TRV was derived from a beta general distribution (NOAEL = 0.0000023 mg/kg and LOAEL = 0.000023 mg/kg). Both sets of TRVs are evaluated in the NBSA fish tissue assessment.

A focused effort was also conducted as part of this BERA to identify and summarize available fish liver TRVs. This included extracting fish liver studies cited in the USACE's Environmental Residue Effects Database (ERED; USACE 2019). Studies reporting liver tissue concentrations and lowest observed adverse effect concentration (LOAEC) (and effects endpoints ranging from the LC10 to LC20) for growth, reproduction, and survival/mortality were included in the datasets used to calculate TRVs. The available fish liver TRV ranges are provided in Table 7-6. Liver-based LOAEC TRVs were available for arsenic, chromium, copper, mercury, selenium, 2,3,7,8-TCDD (used to evaluate 2,3,7,8-TCDD, total dioxin/furan TEQ fish, total PCB TEQ fish, and total dioxin/furan/PCB TEQ fish), and Aroclor 1254 (used to evaluate total PCBs). For COPECs without available LOAEC liver TRVs from USACE (2019), the no observed adverse effects concentration (NOAEC) liver TRVs available in USACE (2019) were included in the evaluation. These included lead, silver, and zinc. The mean NOAEC was selected for the HQ assessment for these metals.

Table 7-6. TRVs for White Perch Liver Assessment

Constituent	Range of Fish Liver TRVs				Selected Liver-based TRV[a]			
	No. Studies	Minimum	Maximum	Mean	NOAEL	LOAEL	Notes (endpoint/ species)	Source
Dioxins/Furans								
2,3,7,8-TCDD	3	0.00371	0.00371	0.00371	--	0.00371	Growth/mortality; rainbow trout	Branson et al. 1985
Total Dioxin/Furan TEQ Fish [b]	3	0.00371	0.00371	0.00371	--	0.00371	Growth/mortality; rainbow trout	Branson et al. 1985
PCBs								
Total PCBs [c]	1	7.88	7.88	7.88	--	7.88	Reproduction; Atlantic cod	Sangalang et al. 1981
Total PCB TEQ Fish [c]	3	0.00371	0.00371	0.00371	--	0.00371	Growth/mortality; rainbow trout	Branson et al. 1985
Total Dioxin/Furan /PCB TEQ Fish [b]	3	0.00371	0.00371	0.00371	--	0.00371	Growth/mortality; rainbow trout	Branson et al. 1985
Pesticides								
Total DDx (2,4 & 4,4)	--	--	--	--	--	--	--	--
Dieldrin	--	--	--	--	--	--	--	--
Total Chlordane	--	--	--	--	--	--	--	--
Inorganics								
Arsenic	2	9.6	18	13.8	--	9.6	Mortality; Mozambique tilapia	Liao et al. 2004
Cadmium	9	1.3	252	29.9	--	1.3	Mortality; dab	von Westernhagen et al. 1980
Chromium	1	0.42	0.42	0.42	--	0.42	Mortality; chinook salmon	Farag et al. 2006
Copper	9	0.84	288	137	--	0.84	Growth; channel catfish	Murai et al. 1981
Lead[a]	15	1.0	20	11.1	11.1	--	Mortality; goldfish	Zhang et al. 2005
Mercury	1	3.0	3.0	3.0	--	3.0	Growth; climbing perch	Panigrahi and Misra 1978
Methylmercury [d]	1	3.0	3.0	3.0	--	3.0	Growth; climbing perch	Panigrahi and Misra 1978
Selenium	8	5.2	20	11.5	--	5.2	Growth; white sturgeon	Tashjian et al. 2006
Silver[a]	2	19	19	19	19	--	Growth/mortality; rainbow trout	Galvez et al. 2001
Zinc[a]	13	3.4	68	35	35	--	Mortality; goldfish	Zhang et al. 2005

Notes:
Units are milligrams per kilogram, wet weight
[a] Liver TRVs based on the LOAEC were not available for lead, silver, and zinc; therefore, liver TRVs based on the NOAEC are provided and used for the evaluation.
[b] Liver TRV based on 2,3,7,8-TCDD.
[c] Liver TRV based on Aroclor 1254.
[d] Liver TRV based on mercury.

~~8.2.3~~7.2.3 Risk Characterization

~~8.2.3.1~~7.2.3.1 Whole-Body Tissue

This section presents the comparison of fish tissue EPCs to NBSA-selected and LPR FFS TRVs to calculate HQs for each COPEC and species. Fish tissue NOAEL and LOAEL HQs are presented in Table 7-7a (NBSA-wide), Table 7-7b (North Zone), Table 7-7c (Southeast Zone), and Table 7-7d (Southwest Zone). The HQs are based on UCLs, or maximum concentrations if there were fewer than six detected values. A summary of COPECs that exceed their TRVs (i.e., HQs that equal or exceed 1) is provided in Table 7-8.

Table 7-8. Summary of TRV Exceedances for Fish Tissue Assessment

Preliminary COCs	NBSA and LPR FFS NOAEL and LOAEL-based HQs ≥ 1.0															
	Mummichog/Killifish								American Eel							
	NBSA-wide		North		SE		SW		NBSA-wide		North		SE		SW	
	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS
Dioxins/Furans	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
2,3,7,8-TCDD		●	-	●	-	●	-	●		●	-	●	-	●	-	●
Total Dioxin/Furan TEQ Fish		●	-	●	-	●	-	●		●	-	●	-	●	-	●
PCBs	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Total PCBs	○	●	○	●	○	●	○	●	○	●	-	●	-	●	○	●
Total PCB TEQ Fish		●	-	○	-	○	-	●		○	-	●	-	-	-	○
Total Dioxin/Furan/ PCB TEQ Fish		●	-	●	-	●	-	●		●	-	●	-	●	-	●
Pesticides	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Total DDx (2,4 & 4,4)		○	-	○	-	○	-	●		○	-	○	-	○	-	●
Dieldrin		○	-	○	-	○	-	-		○	-	○	-	○	-	○
Total Chlordane	○	-	○	-	○	-	○	-	○	-	○	-	○	-	○	-
Inorganics	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Arsenic		-	○	-	-	-	-	-		-	-	-	-	-	-	-
Chromium		-	-	-	-	-	-	-		-	-	-	-	-	-	-
Copper**	●	●	●	●	-	●	○	●		○	-	○	-	○	-	●
Lead		○	-	○	-	○	-	○			-	○	-	-	-	○
Mercury	○		○	○	○	-	○	-	○	○	○	●	○	○	○	○
Methylmercury	○	○	○	○	○	-	○	○	○	○	○	●	○	○	○	●
Selenium*	○	-	○	-	○	-	○	-	○	-	○	-	○	-	○	-
Silver*	○	-	●	-	-	-	-	-		-	-	-	-	-	-	-

Preliminary COCs	NBSA and LPR FFS NOAEL and LOAEL-based HQs ≥ 1.0															
	Mummichog/ Killifish								American Eel							
	NBSA-wide		North		SE		SW		NBSA-wide		North		SE		SW	
	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS
Dioxins/Furans																
2,3,7,8-TCDD	○	●	○	●	○	●	○	●		●	○	●		●		●
Total Dioxin/Furan TEQ Fish	○	●	○	●	○	●	○	●		●	○	●		●		●
Polychlorinated Biphenyls																
Total PCBs	○	●	○	●	○	●	○	●	○	●	○	●	○	●	○	●
Total PCB TEQ Fish		●		○		○		●		○		●				○
Total Dioxin/Furan/PCB TEQ Fish	○	●	○	●	○	●	○	●		●	○	●		●	○	●
Pesticides																
Total DDx (2,4 & 4,4)	○	○	○	○	○	○	○	●	○	○	○	○	○	○	○	●
Dieldrin		○		○		○				○		○		○		○
Total Chlordane		--		--		--		--		--		--		--		--
Hexachlorobenzene	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Inorganics																
Arsenic		--	○	--		--		--		--		--		--		--
Chromium		--		--		--		--		--		--		--		--
Copper	●	●	●	●		●	○	●		○		○		○		●
Lead		○		○		○		○				○				○
Mercury	○		○	○			○		○	○	○	●	○	○	○	○
Methylmercury	○	○	○	○	○		○	○	○	○	●	●	○	○	○	●
Selenium	○	--	○	--	○	--	○	--	○	--	○	--	●	--	○	--
Silver	○	--	●	--		--		--		--		--		--		--
Zinc		--		--		--		--		--		--		--		--

Notes:
● = indicates LOAEL HQ ≥ 1; ○ = indicates NOAEL HQ ≥ 1~~;~~
- = not evaluated in the LPR FFS; COPECs evaluated that did not have any exceedances are not shown.
*Sediment COPEC concentrations are within background levels and tissue residues are assumed to be related to background conditions as well.
~~*Sediment COPEC concentrations are within background levels and tissue residues are assumed to be related to background conditions as well.~~** Tissue residues in mummichog were within the range of Jamaica Bay background residues; all fish tissue copper residues also assumed to be background-related.

Table 7-8. Summary of TRV Exceedances for Fish Tissue Assessment (cont.)

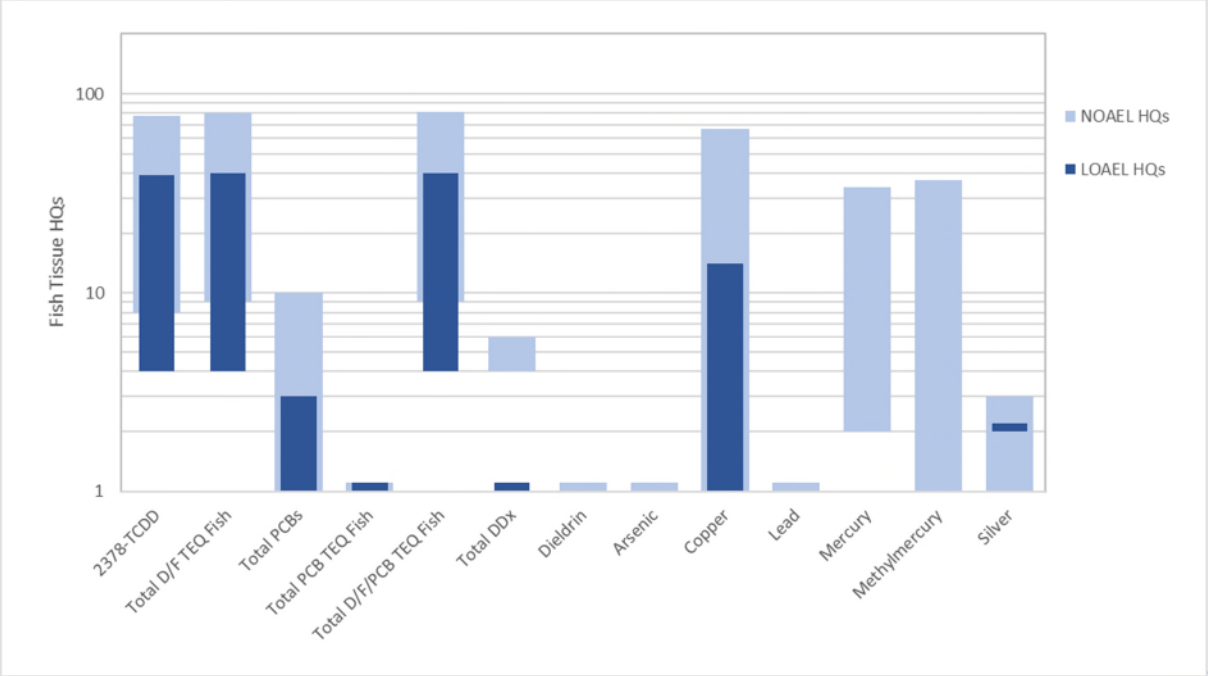
Preliminary COCs	NBSA and LPR FFS NOAEL and LOAEL-based HQs ≥ 1.0															
	Summer/Winter Flounder								White Perch							
	NBSA-wide		North		SE		SW		NBSA-wide		North		SE		SW	
	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS
Dioxins/Furans	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
2,3,7,8-TCDD		●	-	●	-	●	-	●		●	○	●	-	●	-	●
Total Dioxin/Furan TEQ Fish		●	-	●	-	●	-	●		●	○	●	-	●	-	●
PCBs	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Total PCBs		○	-	●	-	○	-	○	○	●	○	●	○	●	○	●
Total PCB TEQ Fish			-	-	-	-	-	-		○	-	●	-	○	-	○
Total Dioxin/Furan/ PCB TEQ Fish		●	-	●	-	●	-	●		●	○	●	-	●	-	●
Pesticides	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Total DDx (2,4 & 4,4)			-	-	-	-	-	-		●	-	●	-	●	-	●
Dieldrin			-	-	-	-	-	-		○	-	○	-	○	-	○
Total Chlordane	○	-	○	-	○	-	○	-	○	-	○	-	○	-	○	-
Inorganics	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Arsenic	○	-	○	-	○	-	○	-		-	-	-	-	-	-	-
Chromium		-	-	-	-	-	-	-		-	●	-	-	-	-	-
Copper**		○	-	●	-	○	-	○	●	●	●	●	●	●	●	●
Lead		○	-	○	-	○	-	-			-	-	-	-	-	-
Mercury	○	○	○	○	○	○	○	○	○	○	○	○	○	○	○	○
Methylmercury	○	○	○	○	○	○	○	○	○	○	○	○	○	○	○	○
Selenium*	○	-	○	-	○	-	○	-	○	-	○	-	○	-	○	-
Silver*		-	-	-	-	-	-	-	●	-	●	-	●	-	●	-

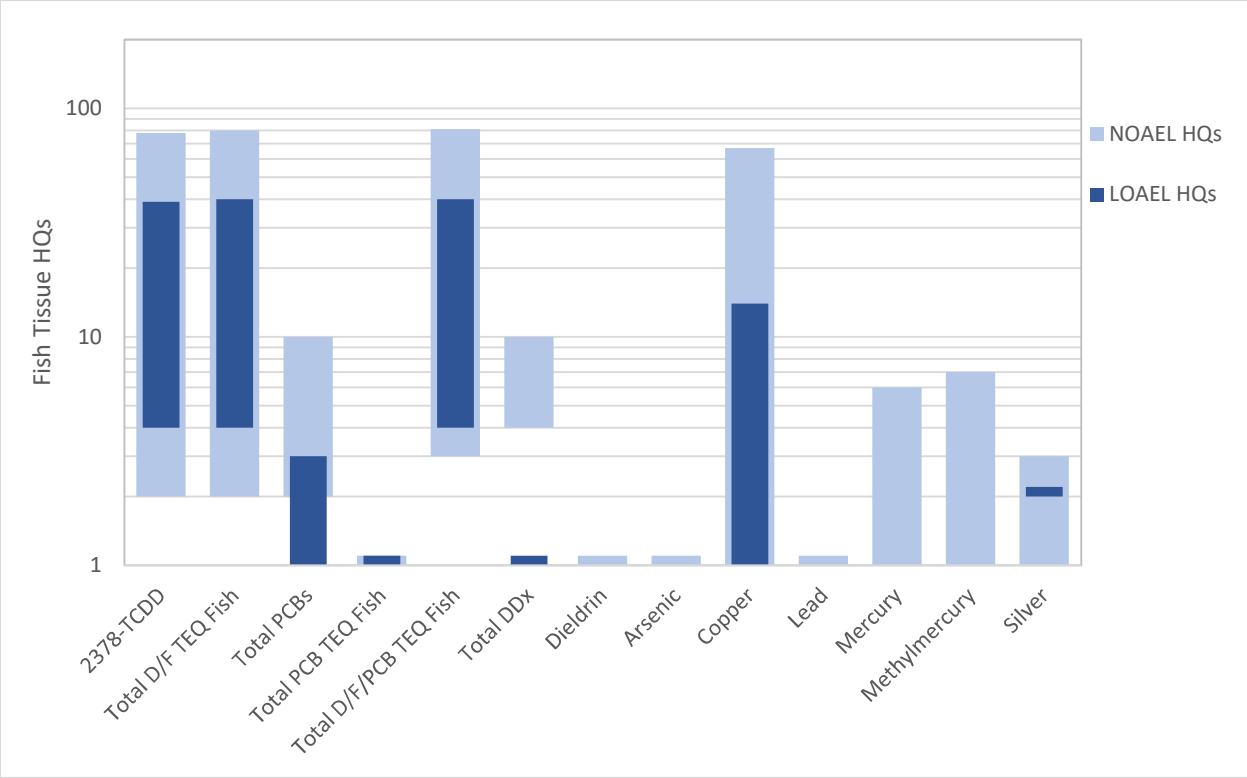
Preliminary COCs	NBSA and LPR FFS NOAEL and LOAEL-based HQs ≥ 1.0															
	Summer/Winter Flounder								White Perch							
	NBSA-wide		North		SE		SW		NBSA-wide		North		SE		SW	
	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS
Dioxins/Furans																
2,3,7,8-TCDD		●	○	●		●		●	○	●	●	●	○	●	○	●
Total Dioxin/Furan TEQ Fish		●	○	●		●		●	○	●	●	●	○	●	○	●
Polychlorinated Biphenyls																
Total PCBs		○	○	●		○		○	○	●	○	●	○	●	○	●
Total PCB TEQ Fish										○		●		○		○
Total Dioxin/Furan/PCB TEQ Fish		●	○	●		●		●	○	●	●	●	○	●	○	●
Pesticides																
Total DDx (2,4 & 4,4)									●	●	○	●	○	●	●	●
Dieldrin										○		○		○		○
Total Chlordane		--		--		--		--		--		--		--		--
Hexachlorobenzene	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
Inorganics																
Arsenic	○	--	○	--	○	--	○	--		--		--		--		--
Chromium		--		--		--		--		--	●	--		--		--
Copper		○		●		○		○	●	●	●	●	●	●	●	●
Lead		○		○		○										
Mercury	○	○	○	○	○	○	○	○	○	○	○	○	○	○	○	○
Methylmercury	○	○	○	○	○	○	○	○	○	○	○	○	○	○	○	○
Selenium	○	--	○	--	○	--	○	--	○	--	○	--	○	--	●	--
Silver		--		--		--		--	●	--	●	--	●	--	●	--
Zinc		--		--		--		--		--		--		--		--

Notes:
● = indicates LOAEL HQ ≥ 1~~;~~
○ = indicates NOAEL HQ ≥ 1~~;~~
- = not evaluated in the LPR FFS; COPECs evaluated that did not have any exceedances are not shown. ~~*Sediment COPEC concentrations are within background levels and tissue residues are assumed to be related to background conditions as well.~~
*Sediment COPEC concentrations are within background levels and tissue residues are assumed to be related to background conditions as well.
** Tissue residues in mummichog were within the range of Jamaica Bay background residues; all fish tissue copper residues also assumed to be background-related.

The range of HQs for the aggregate whole-body fish tissue samples (i.e., all species) with HQs ≥ 1 are depicted in Figure 7-1. ~~Arsenic, lead, selenium, zinc, dieldrin, and total chlordane resulted in~~NBSA-wide LOAEL HQs ~~less than~~were ≥ 1 for ~~all~~2,3,7,8-TCDD, total dioxin/furan TEQ, total PCBs, total PCB TEQ, total dioxin/furan/PCB TEQ, total DDx, and some inorganics (primarily copper) for most receptors using ~~both~~ the ~~NBSA- and~~ LPR FFS~~-selected~~ TRVs. ~~Hexachlorobenzene and zinc concentrations were less than both NOAEL and~~NBSA-wide LOAEL HQs were ≥ 1, total DDx, copper, and silver using the NBSA TRVs for ~~all fish.~~some receptors.

Figure 7-1. Magnitude of NBSA-wide NOAEL and LOAEL HQs ≥ 1 for Fish Tissue Assessment





~~8.2.3.2~~7.2.3.2 White Perch Liver Tissue

In addition to whole-body tissue, several white perch liver tissue samples were collected to provide additional perspective on the uptake and processing of COPECs. In fish and other vertebrates, the liver functions to filter waste from the blood and serves as the primary site for *in vivo* metabolism and/or excretion of exogenous and endogenous chemicals. The liver is centrally located in the fish circulatory system and lacks a basement membrane, which increases exchange between the blood and hepatic tissue for chemicals present in fish via both dietary and brachial (i.e., via the gills) exposure routes (Di Giulio and Hinton 2008). As a result, concentrations of COPECs are typically higher in liver than in other tissues.

White perch was selected for this evaluation because it is: 1) a species that is known to accumulate higher levels of contaminants than other fish (based on findings from the LPR [see Windward 2019]), and 2) a species for which enough individuals could be collected in the NBSA during a focused sampling period to composite the requisite liver mass to analyze for the full suite of constituents identified in Table 4-2.

Liver tissue EPCs were compared to the range of available TRVs to calculate liver tissue HQs. Consistent with the approach for TRV derivation used in this BERA, lowest available liver tissue LOAEL was compared to the results of the white perch liver samples, and the resulting HQs are presented in Table 7-9. Liver tissue EPCs are based on UCLs, or maximum concentrations if there were fewer than six detected values. For liver, the North, Southeast, and Southwest Zone EPCs are based on maximum detected concentrations due to the low sample sizes in each zone.

Table 7-9. Calculated White Perch Liver Tissue HQs

Constituent	White Perch Liver (Liver-based TRVs)			
	HQs based on the Minimum Liver TRV			
	NBSA-wide	North	Southeast	Southwest
Dioxins/Furans				
2,3,7,8-TCDD	0.02	0.03	0.02	0.02
Total Dioxin/Furan TEQ Fish	0.02	0.03	0.02	0.02
PCBs				
Total PCBs	0.3	0.4	0.2	0.3
Total PCB TEQ Fish	~~1E-03~~0.001	~~2E-03~~0.002	~~6E-04~~0.0006	~~1E-03~~0.001
Total Dioxin/Furan/ PCB TEQ Fish	~~2E-~~0.02	~~3E-02~~0.03	~~2E-~~0.02	~~2E-~~0.02
Pesticides				
Total DDx (2,4 & 4,4)	-	-	-	-
Dieldrin	-	-	-	-
Total Chlordane	-	-	-	-
Inorganics				
Arsenic	**1**	0.1	0.1	**2**
Chromium	**12**	0.4	0.3	**12**
Copper**	**4,312**	**1,175**	**1,190**	**9,286**

Lead	0.~~01~~1	0.~~01~~1	0.~~01~~1	0.~~02~~2
Mercury	0.08	0.09	0.07	0.08
Methylmercury	0.03	0.04	0.04	0.03
Selenium*	**18**	**6**	**5**	**39**
Silver*	**4**	**1**	**1**	**8**
Zinc	**~~5~~52**	**~~2~~23**	**~~2~~20**	**~~12~~121**

Notes:
"-" = Not evaluated in the LPR FFS (whole-body TRVs), or not available in the Environmental Residue Effects Database (liver TRVs).
*Sediment COPEC concentrations are within background levels and tissue residues are assumed to be related to background conditions as well.
**Tissue residues in mummichog were within the range of Jamaica Bay background residues; white perch residues are also assumed to be background-related.

When comparing the liver EPC to liver TRVs, HQs for dioxins/furans, total PCBs, lead, mercury, and methylmercury are less than 1. For arsenic, chromium, copper, and selenium, HQs are greater than 1 in the NBSA-wide area and one or more individual assessment zones. The HQs for copper are particularly elevated, but this is due to the wide range (i.e., three orders of magnitude) of reported liver TRVs, and the use of the lowest of the TRVs in this assessment. If the mean of the reported TRVs were applied, the resulting HQs would be above 1, but in the same range as those for whole body tissues.

Liver TRVs were not available from the Environmental Residue Effects Database for any pesticides (total DDx, dieldrin, total chlordane), and only NOAEC liver TRVs are available for lead, silver, and zinc. The HQs were greater than 1 for silver and zinc, but not for lead.

For selenium and silver, sediment concentrations in the NBSA are at or below background concentrations in Jamaica Bay sediments (Table 4-14). Although these metals were measured in white perch liver and other fish tissues, they likely reflect background exposures. In particular, selenium is a critical nutrient in fish. The HQs greater than 1 estimated for selenium and silver for fish are likely a result of uncertainties in the toxicity values for these metals. The selenium and silver NOAEC HQs ≥ 1 are not considered to indicate that adverse effects to fish are occurring. Similarly, while the zinc NOAEC liver HQs are > 1, there is uncertainty associated with both the exposure and effects assumptions. The liver datasets are smaller than for whole body tissue; only the NBSA-wide liver EPC is based on a 95UCL, as the assessment zones had too few samples to calculate this statistic. Additionally, there is a high degree of uncertainty in the interpretation of liver HQs to effects in whole fish and fish populations. Specifically, for zinc, fish actively regulate the uptake and distribution of this essential metal (Bury et al. 2003) and tissue burdens vary widely between species (USEPA 2007b). Although there is uncertainty in the use of tissue HQs for metals, the whole-body TRVs are considered to be less uncertain for estimating risk to individuals and fish populations than liver TRVs. The fact that zinc did not exceed any of its NOAEL or LOAEL HQs in whole-body sample of any fish species evaluated indicates that it does not pose a risk. It is interesting to note that the highest liver HQs calculated for metals consistently occurs in the Southwest Zone. This is consistent with the findings of elevated metals concentrations in sediments and whole-body tissues of invertebrates and some fish from this zone.

~~8.3~~7.3 Fish Dietary Assessment

The dietary assessment was conducted for the following species: mummichog/killifish (forage fish), summer/winter flounder (benthic omnivore), white perch (pelagic invertivore), and American eel (pelagic piscivore). The reason for conducting the dietary assessment was to evaluate potential risks from COPECs that may be taken up by fish through their diet, but are physiologically regulated, and do not necessarily bioaccumulate in tissues. This includes PAHs (which are readily metabolized by fish and excreted as metabolites of the parent compounds), and inorganic chemicals. For these COPECs, the bioaccumulation-based tissue assessment (Section 7.2) may not adequately represent exposure-risks. Therefore, a focused dietary assessment was conducted as a secondary LOE for these COPECs to evaluate whether daily doses ingested by fish may pose risk.

This section describes how exposure and effects concentrations were derived, and presents the HQs for the fish dietary assessment. The COPECs evaluated for the fish dietary assessment are limited to PAHs and inorganics (arsenic, cadmium, chromium, copper, lead, mercury/methylmercury, selenium, silver, and zinc).

~~8.3.1~~7.3.1 Exposure

This section presents the methods for calculating exposure doses, including descriptions of the selection of parameters for the dose equation, composition of prey in diet, exposure areas, and EPCs in prey.

~~8.3.1.1~~7.3.1.1 Methods

Dietary doses for fish were estimated based on ingestion of biota (i.e., prey) and the incidental ingestion of sediment. Dietary doses were estimated as milligrams of each COPEC ingested per kilogram of body weight per day using the following equation:

$$ADD_{total} = [(EPC_s * SIR/BW * DF_s) + \Sigma(EPC_k * FIR_k/BW * DF_k)] * SUF$$ ***Equation 7-1***

where,

ADD_{total} = average daily dose in milligrams per kilogram body weight per day (mg/kg bw/day)

EPC_s = exposure point concentration in sediment (milligrams per kilogram [mg/kg], dry weight [dw])

SIR = ingestion rate for sediment (kg/day, dw)

BW = body weight of organism (kg)

DF_s = dietary fraction of sediment ingested (range 0 to 1)

EPC_k = EPC for the kth food type (mg/kg/ day, wet weight [ww])

FIR_k = ingestion rate for the kth food type ((kg/day, ww)

DF_k = dietary fraction of intake of the kth food type (range 0 to 1)

SUF = site use factor (unitless); proportion of time the selected species spends foraging in the NBSA

~~8.3.1.2~~7.3.1.2 Body Weights, Ingestion Rates, Prey Composition, Site Use Factors, Exposure Areas

The description of the exposure factors for the various fish receptors is provided in Appendix B. Body weights for each species were based on the average of all body weights for individuals of that particular species collected during the 2014, 2015, and 2016 NBSA fish sampling events. Ingestion rates were obtained from the literature for each representative species being modelled. Body weights and ingestion rates for each receptor are presented in Table 7-10.

The dietary fraction of each component in each species' diet is based on information obtained from the literature (reviewed in Appendix B) and are presented in Table 7-10. Prey ingested by fish species were limited to whole-body samples only from prey types for which empirical tissue chemistry data from the NBSA were available. Fish were separated by length into two groups: fish ≤ 15 cm and fish ≤ 30 cm.

Several studies conducted in varied habitats have found that the mummichog diet consists of detritus, algae, small crustaceans (i.e., amphipods, tanaids, copepods, and ostracods), insects (adult and larvae), and polychaetes (Abraham 1985). Polychaete worm tissue data from the bioaccumulation study were used to represent the benthic prey portion in the mummichog diet (i.e., 100% of the mummichog diet was based on worm tissue).

Adult summer and winter flounder feed mostly on fish and crustaceans as well as squid, polychaetes, and shrimp (Packer et al. 1999). Their diet is related to size: the larger the individual, the greater the size of the food item to be consumed. A diet of 25% polychaete worms, 25% blue crabs, 25% softshell clam, and 25% fish (≤ 30 cm) was used to represent the prey consumed by flounder evaluated in the NBSA.

White perch are invertivorous benthic-feeding fish; common dietary components include amphipods, shrimp, copepods, and fish, based on regional studies for the Hudson and Hackensack Rivers (Bath and O'Connor 1985; Weis 2005). Blue crab (a surrogate for shrimp) and small fish (< 15 cm) were assumed to each make up 20% of the diet. Softshell clam (a surrogate for copepods and amphipods) were assumed to make up 20% of the diet. NBSA polychaete worms were used to represent the remaining portion (40%) of the white perch diet (a surrogate for amphipods and unidentified material).

American eel are opportunistic carnivores and have a diverse diet that includes benthic invertebrates (e.g., insects, crayfish, snails, worms), gastropods, and fish (Atlantic States Marine Fisheries Commission [ASMFC] 2000). As eel grow, a clear shift occurs in the percent composition of prey in their diet; larger eel have a diet of mainly fish and crustaceans, while smaller eel mainly prey on insects (ASMFC 2000). Small eel (< 50 cm) were assumed to consume approximately 40% worms (surrogate for insect species), 20% blue crabs (surrogate for crustaceans), 20% softshell clams (surrogate for gastropods), and 20% fish (≤ 15 cm). Larger eel (≥ 50 cm) were assumed to consume approximately 20% worms (surrogate for insect species), 20% blue crab (surrogate for crustaceans), 20% softshell clam (surrogate for gastropods), and 40% fish (≤ 30 cm).

While fish, blue crab, and softshell clam data were field collected, the worm tissue data were based on the laboratory bioaccumulation study in which worms were exposed to homogenized sediment collected from the 0 to 15-cm depth horizon.

The SUFs for the representative fish receptors are provided in Table 7-10. Mummichog are considered resident in Newark Bay. American eel spawning occurs in the Sargasso Sea and then juvenile eels migrate into coastal waters with some moving into freshwater streams, but some remaining in coastal estuaries and bays until maturity (ASMFC 2000, 2009). Sampling data indicate American eel are present in the NBSA year-round. Other species are migratory and only use the site for a portion of each year. Summer flounder are found in bays and estuaries from late spring to early autumn, when they migrate offshore to the outer continental shelf; winter flounder undertake small-scale migrations into estuaries and embayments in winter to spawn, then move to deeper water during summer (Grimes et al. 1989). White perch are primarily found in low salinity waters upstream of Newark Bay, but are common in Newark Bay in winter and when freshwater runoff produces low salinity conditions (USACE 1997).

The exposure areas assessed for the receptor fish species are provided in Table 3-3. Mummichog risk from prey consumption was evaluated NBSA-wide and by assessment zone (north, southeast, southwest) and risk from incidental sediment ingestion was evaluated using sediment samples from NBSA-wide mudflats and mudflats within assessment zones. Risk (from prey and incidental sediment ingestion) to summer/winter flounder, white perch, and American eel were evaluated NBSA-wide and by assessment zone. All of the fish receptor species included polychaete worm as a portion of their diet; however, polychaete worm sample size was not large enough to be evaluated by assessment zone. Therefore, risk from ingestion of polychaete worms were assessed NBSA-wide in the risk calculations for fish. See Table 4-10 and Figure 4-4 for sediment samples assigned to each assessment zone and mudflats, and Table 4-11 for tissue samples assigned to each assessment zone.

~~8.3.1.3~~7.3.1.3 Exposure Point Concentrations

EPCs were calculated for each of the two modeled media types (prey and sediment) to calculate dietary doses using Equation 7-1. For prey concentrations, EPCs were calculated separately for each of the prey types for each species (worms, blue crab, softshell clam, fish ≤ 15 cm, and fish ≤ 30 cm) as the UCL for each prey group. The EPCs for prey were calculated NBSA-wide and by assessment zone (except polychaete worm, where EPCs were calculated NBSA-wide only due to limited sample size). For sediment, the EPCs were equal to the UCLs using data from the relevant exposure areas for each fish species. UCL concentrations were calculated using USEPA's ProUCL® statistical package (Version 5.1) (USEPA 2015b).[7] For each dataset with fewer than six detected samples or fewer than six total samples, a UCL was not calculated; instead, the maximum detected concentration was used as the EPC. If no samples were detected, then the maximum non-detect value was used as the EPC. The tissue data used in the fish dietary assessment are summarized in ~~Table~~Tables 4-~~11.~~7d-k The EPCs for the fish dietary assessment are presented in Appendix C.

~~8.3.1.4~~7.3.1.4 Estimated Doses

Dietary doses were calculated based on Equation 7-1 using the prey and sediment ingestion rates and species body weights and prey composition from Table 7-10. The EPCs are presented in Appendix C.

[7] The UCL recommended by USEPA's ProUCL® software (typically the 95% UCL, but in some cases the 97.5% or even the 99% UCL) was used. The selected UCL statistic is presented in Appendix C-1.

~~8.3.2~~7.3.2 Effects

Dietary fish TRVs for the COPECs identified in Table 3-5 are provided in Table 7-11, and discussed in detail in Appendix D. The selected TRVs were primarily derived from the LPRSA BERA TRV database (Windward 2019~~).~~) and are consistent with the Final LPRSA BERA TRVs for COPECs common to both areas (NBSA and LPRSA). For the LPRSA BERA Windward (2019) conducted a literature search for relevant toxicological studies, and derived acceptable NOAELs and LOAELs. The lowest acceptable LOAEL and highest NOAEL below the LOAEL from the same study ~~based on marine organisms were chosen. If marine TRVs were not available, then the freshwater-based TRV selected by LPRSA BERA was used.~~was chosen. Several COPECs were not evaluated in the LPRSA BERA fish dietary risk assessment including arsenic, lead, and silver. TRVs for arsenic and silver were available in the LPRSA BERA TRV database (Windward 2019). A lead TRV based on Mount et al. (1994) was selected. No fish dietary TRVs were derived for the USEPA LPR FFS. The selected NBSA TRVs for these additional COPECs are further discussed in Appendix D.

~~8.3.3~~7.3.3 Risk Characterization

This section presents the comparison of fish dietary doses to NBSA-selected TRVs to calculate HQs for each COPEC and species. Fish NOAEL and LOAEL HQs based on dietary doses for mummichog/killifish, American eel, summer/winter flounder, and white perch are presented in Table 7-12a (NBSA-wide), Table 7-12b (North Zone), Table 7-12c (Southeast Zone), and Table 7-12d (Southwest Zone). The HQs are based on UCLs, or maximum concentrations if there were fewer than six detected values. Appendix E provides dietary doses, TRVs, and calculated HQs for the fish dietary COPECs. A summary of COPECs that exceed their dietary TRVs (i.e., HQs that equal or exceed 1) in fish is provided in Table 7-13.

Table 7-13. Summary of TRV Exceedances for Fish Dietary Assessment

~~Preliminary COCs~~	~~NBSA NOAEL and LOAEL-based HQs ≥ 1.0~~							
	~~Mummichog~~				~~American Eel (All Sizes)~~			
	~~NBSA-wide~~	~~North~~	~~South east~~	~~South west~~	~~NBSA-wide~~	~~North~~	~~South east~~	~~South west~~
~~Inorganics~~	-	-	-	-	-	-	-	-
~~Cadmium~~	~~●~~	~~●~~	~~●~~	~~●~~	~~●~~	~~●~~	~~●~~	~~●~~
~~Copper~~	-	-	-	-	-	-	-	~~●~~

	~~NBSA NOAEL and LOAEL-based HQs ≥ 1.0~~

~~Preliminary COCs~~	~~Summer/Winter Flounder~~				~~White Perch~~			
	~~NBSA-wide~~	~~North~~	~~South east~~	~~South west~~	~~NBSA-wide~~	~~North~~	~~South east~~	~~South west~~
~~Inorganics~~	-	-	-	-	-	-	-	-
~~Cadmium~~	●	○	●	●	●	●	●	●
~~Copper~~				○	-	-	-	-

Preliminary COCs	NBSA NOAEL and LOAEL-based HQs ≥ 1.0											
	Mummichog				American Eel <50 cm				American Eel ≥50 cm			
	NBSA-wide	North	SE	SW	NBSA-wide	North	SE	SW	NBSA-wide	North	SE	SW
	NBSA	NBSA	NBSA	NBSA	NBSA	NBSA	NBSA	NBSA	NBSA	NBSA	NBSA	NBSA
Inorganics												
Cadmium	●	●	●	●	●	●	●	●	●	●	●	●
Copper								○				○
Mercury	○	○	○	●	●	●	●	●	●	●	●	●
Methylmercury	○	○	○	○	○	○	●	○	●	●	●	●
Selenium	○	○	○	○	○	○	○	○	○	○	○	○

Preliminary COCs	NBSA NOAEL and LOAEL-based HQs ≥ 1.0							
	Summer/Winter Flounder				White Perch			
	NBSA-wide	North	SE	SW	NBSA-wide	North	SE	SW
	NBSA	NBSA	NBSA	NBSA	NBSA	NBSA	NBSA	NBSA
Inorganics								
Cadmium	●	○	●	●	●	●	●	●
Copper				○				
Mercury	●	●	○	●	○	●	○	○
Methylmercury	○	○	○	○	○	○	○	○
Selenium	○	○	○	○	○	○	○	○

Notes:
● = indicates LOAEL HQ ≥ 1~~;~~
○ = indicates NOAEL HQ ≥ 1~~.~~
Fish dietary HQs not evaluated in the LPR FFS. LOAEL is not available for chromium, lead, and silver; therefore, results are presented based on the NOAEL. COPECs evaluated that did not have any exceedances are not shown.

Dietary LOAELs ~~of~~≥ 1 for cadmium were found NBSA-wide and in various zones for all fish species~~.~~ and for mercury and methylmercury for some of the species (primarily American eel < 50 cm and ≥ 50 cm) NBSA-wide and in several of the assessment zones. The remaining metals (i.e., arsenic, chromium, copper, lead~~, mercury, methylmercury~~, nickel, selenium, silver, zinc) and total PAHs all have LOAEL HQs less than 1 for all receptors evaluated~~. Copper exceeded the LPR FFS NOAEL only in one location—the Southwest Zone.~~.

~~8.4~~7.4 Fish Egg Tissue Assessment

Potential risks to fish eggs from maternal dietary exposure were evaluated for mummichog and killifish (i.e., combined *Fundulus* ~~spp.).~~sp.), the only resident fish species in the Bay. In this assessment, fish egg tissue chemical concentrations were estimated using chemical-specific egg-to-adult conversion factors from the literature. ~~The~~Consistent with the evaluation conducted in the LPRSA BERA, the NBSA fish egg risk assessment was limited to the resident *Fundulus* species for a select group of bioaccumulative compounds that are known to pose potential reproductive risks in fish, including dioxins/furans, PCBs, associated TEQs, and mercury/methylmercury.

~~8.4.1~~7.4.1 Exposure

Fish egg tissue chemical concentrations for *Fundulus* ~~spp~~sp. were estimated using egg to adult conversion factors (CFs). The following sections describe CFs for total PCBs, TEQs, and methylmercury/mercury, and present the modeled concentrations based on the CFs.

~~8.4.1.1~~7.4.1.1 CFs for Total PCBs, Dioxin/Furan TEQ, and Dioxin/Furan/PCB TEQs

For organic fish egg COPECs (i.e., total PCBs, dioxin/furan TEQ, PCB dioxin/furan TEQ), CFs were calculated using lipid-normalized concentrations based on data reported by Niimi (1983) and Russell et al. (1999) using the following equation:

$$CF = (C_{egg,lipid})/(C_{adult,lipid})$$ ***Equation 7-2***

where:

CF = adult-to-egg conversion factor

$C_{egg,lipid}$ = lipid-normalized chemical concentration in egg tissue

$C_{adult,lipid}$ = lipid-normalized chemical concentration in adult whole-body tissue

Based on the data reported in Niimi (1983), the following equation was derived to predict egg tissue concentrations from adult concentrations on a lipid-normalized basis for total PCBs, dioxin/furan TEQ, and PCB dioxin/furan TEQ:

$$EPC_{egg,lipid} = 0.6213 \times EPC_{adult,lipid}$$ ***Equation 7-3***

where,

$EPC_{egg,lipid}$ = exposure point concentration in egg tissue (mg/kg-lipid dw)

$EPC_{adult,lipid}$ = exposure point concentration in adult whole-body tissue (mg/kg-lipid dw)

Based on the data reported in Russell et al. (1999), ~~the~~a CF of 1 was also evaluated. The following equation was derived to predict egg tissue concentrations from adult concentrations on a lipid-normalized basis for total PCBs, dioxin/furan TEQ, and PCB dioxin/furan TEQ:

$$EPC_{egg,lipid} = EPC_{adult,lipid}$$ ***Equation 7-4***

where,

$EPC_{egg,lipid}$ = exposure point concentration in egg tissue (mg/kg-lipid dw)

$EPC_{adult,lipid}$ = exposure point concentration in adult whole-body tissue (mg/kg-lipid dw)

The UCL lipid value in LPRSA mummichog eggs (3.3%; Windward 2019) was used to convert lipid-normalized egg concentrations to wet weight egg concentrations for comparison to fish egg TRVs.

~~8.4.1.2~~7.4.1.2 CFs for Methylmercury

For methylmercury/mercury, a robust regression model could not be developed based on the data presented in Niimi (1983). Thus, there is high uncertainty in predicting egg tissue concentrations from whole-body tissue concentrations. Mercury wet weight egg to adult CFs from Niimi (1983) ranged from 0.039 to 0.101. The maximum CF of 0.101 was used in the following equation to predict egg tissue concentrations from adult concentrations for methylmercury:

$C_{egg,ww} = 0.101 \times C_{adult,ww}$ ***Equation 7-5***

where,

$C_{egg,ww}$ = chemical concentration in egg tissue (mg/kg-lipid dw)

$C_{adult,ww}$ = chemical concentration in adult whole-body tissue (mg/kg-lipid dw)

A CF of 1 was also evaluated for methylmercury, similar to the evaluation for total PCBs, Dioxin/Furan TEQ, and Dioxin/Furan/PCB TEQs.

~~8.4.1.3~~7.4.1.3 Modeled Egg Concentration

The modeled concentrations in mummichog eggs were estimated using *Fundulus* ~~spp~~sp. (combined adult mummichog and striped killifish) whole-body tissue UCL concentrations[8] and are presented in Appendix C. UCLs for *Fundulus* ~~spp~~sp. tissue EPCs were calculated using USEPA's ProUCL® statistical package (Version 5.1) (USEPA 2015b).[9] Fish egg EPC calculations are detailed in Appendix C.

~~8.4.2~~7.4.2 Effects

Fish egg TRVs for the COPECs identified in Table 3-5 are provided in Table 7-14 and discussed in detail in Appendix D. Fish egg TRVs were primarily derived from the LPRSA BERA TRV database (Windward 2019). In addition, fish egg TRVs from the LPR FFS (USEPA 2014) were also included for comparison with the NBSA-selected TRVs.

~~8.4.3~~7.4.3 Risk Characterization

This section presents the comparison of fish egg EPCs to TRVs to calculate HQs for all COPECs. HQs based on modeled fish egg concentrations and fish egg TRVs presented in Table 7-14 were calculated for the fish egg COPECs and are presented in Table 7-15. Appendix C provides EPCs for the fish egg tissue COPECs. A summary of COPECs that exceed their TRVs (i.e., HQs that equal or exceed 1) is provided in Table 7-16. ~~No fish~~Fish egg concentration estimates exceeded LOAEL TRVs for Total PCBs

[8] Fillet and organ-specific samples were not available and not included in UCL calculations.
[9] The UCL recommended by USEPA's ProUCL® software (typically the 95% UCL, but in some cases the 97.5% or even the 99% UCL) was used. The selected UCL statistic is presented in Appendix C-7.

using both conversion factors (CF = 0.6213 and CF=1) and methylmercury using CF=1 for both NBSA and LPR FFS TRVs.

Table 7-16. Summary of TRV Exceedances for Fish Egg Assessment

~~Constituent~~	~~Fish Egg Tissue - HQs (NBSA-Wide)~~			
	~~Mummichog/Killifish~~			
	~~CF = 0.6213 (Dioxins/Furans/PCBs); CF = 0.101 (Methylmercury)~~		~~CF = 1 (Dioxins/Furans/PCBs)~~	
	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~
~~Dioxins/Furans~~	~~-~~	~~-~~	~~-~~	~~-~~
~~2,3,7,8-TCDD~~	~~○~~	~~○~~	~~○~~	~~○~~
~~Total Dioxin/Furan TEQ Fish~~	~~○~~	~~○~~	~~○~~	~~○~~
~~PCBs~~	~~-~~	~~-~~	~~-~~	~~-~~
~~Total PCBs~~	~~-~~	~~-~~	~~-~~	~~-~~
~~Total PCB TEQ Fish~~	~~-~~	~~-~~	~~-~~	~~-~~
~~Total Dioxin/Furan/PCB TEQ Fish~~	~~○~~	~~○~~	~~○~~	~~○~~
~~Inorganics~~	~~-~~	~~-~~	~~-~~	~~-~~
~~Mercury~~	~~-~~	~~-~~	~~-~~	~~-~~
~~Methylmercury~~	~~○~~	~~-~~	~~-~~	~~-~~

Constituent	Fish Egg Tissue - HQs (NBSA-Wide)			
	Combined Mummichog Killifish			
	CF = 0.6213 (PCBs/Dioxins); CF = 0.101 (Methylmercury)		CF = 1 (all constituents)	
	NBSA	LPR FFS	NBSA	LPR FFS
Dioxins/Furans				
2,3,7,8-TCDD	○	○	○	○
Total Dioxin/Furan TEQ Fish	○	○	○	○
Polychlorinated Biphenyls				
Total PCBs	●	●	●	●
Total PCB TEQ Fish				
Total Dioxin/Furan/PCB TEQ Fish	○	○	○	○
Inorganics				
Mercury			○	○
Methylmercury	○	○	●	●

Notes: ● = indicates LOAEL HQ ≥ 1; ○ = indicates NOAEL HQ ≥ 1. **-** = Not calculated

~~8.5~~7.5 Fish Pathology Findings/Observations

Fish pathology evaluations were conducted on fish collected in the 2014 and 2015 fish sampling program. Fish were collected from various locations along the intertidal mudflats, subtidal flats, transitional slopes, and navigational channel throughout each of the three zones (north, central, south) in the NBSA. Fish were collected from each of the different geographic areas to provide spatial coverage across the entire NBSA.

The pathology examinations were conducted in accordance with standard operating protocols developed by Dr. Jeffrey Wolf, DVM, ACVP, a board-certified veterinary senior pathologist with Experimental Pathology Laboratories, Inc. (EPL) in Sterling, VA. In accordance with the Fish QAPP (Tierra 2014b), the first five fish obtained for each species (unless they were kept for chemical analysis) and 20 fish of two species (scup and summer flounder) were examined. In some cases, five fish of one species were not collected while the pathologist was on site, so a lower number were examined. For the forage fish, a subset of individual fish was externally examined. In addition, a representative sample of individual non-forage fish (up to at least 20 individuals for each of two different species) were subjected to internal pathological examination. Examinations took place within hours of capture.

External examinations included the general body condition (body form), body surfaces, lips and jaws, snout, barbels (where present), operculum (opercle), isthmus, eyes, fins, gills, pseudobranch, branchial cavity, and urogenital opening. Any gross physical abnormalities, lesions, or anomalies were documented. The internal examination included overall morphology, gonad condition, presence of lesions, gonadosomatic index, and internal physical conditions/abnormalities. The fish pathology reports are provided as Attachment 5 to this BERA, and a summary of the pathology findings from this investigation are presented in Table 7-17.

In a 5-day consecutive period (October 6 through 10, 2014) during the fall 2014 fish sampling event, a total of 122 fish representing 26 distinct species were necropsied. Gross external and internal examinations revealed abnormal findings in 11 of 122 (9%) fish. Most findings were of mild severity and likely of minimal pathological significance. In five of 11 cases, findings consisted of external or internal parasite infections. Other findings included fin or skin erosions (N=3), pale nodular areas in the heart (N=2), and an enlarged and discolored ovary (N=1).

On July 27, 2015, an additional 23 fish representing five distinct species were necropsied. Gross external and internal examinations revealed abnormal findings in 1 of 23 fish. One dermal ulcer was found on an Atlantic tomcod. The results from this effort show that there was a very low incidence of minor pathological significance.

Overall, abnormal findings were observed in 12 of the 145 fish (8%) collected and necropsied during the two sampling events. According to Dr. Wolf, the fish pathology investigation showed “only low percentage [of] incidences of relatively minor pathologies across this spectrum of [representative] species” that occur in the NBSA (forage, predatory, benthic, demersal, and pelagic species; resident and migratory species; and adult and juvenile fish) (EPL 2016a).

The fish pathology results do not suggest any health impairments in the fish populations utilizing the NBSA. Although gross health examinations have limited ability to assess potential sublethal (i.e., growth and reproductive) effects, and cannot identify potential risk associated with specific COPECs in

environmental media, the external and internal examinations provide an important line of evidence that speaks to the overall general good health of the fish.

~~8.6~~7.6 Uncertainties Assessment

The following is an uncertainties assessment for the fish risk assessment. Both quantitative and qualitative characterizations are provided for the tissue, dietary, and egg assessments presented in Sections 7.2, 7.3, and 7.4, respectively. The objective of the uncertainty assessment is to provide perspective on the likely accuracy of the overall risk conclusions that are presented in Section 7.7.

For the fish risk assessment, sources of uncertainty that influenced fish risk characterization included uncertainties in the analytical results, data evaluation, CSM, exposure assessment, effects assessment, and interpretation of the risk estimates. Many of these sources of uncertainty are generic in nature and inherent in the risk assessment process, and particularly with reliance on literature-based TRVs and the computation of HQs to make conclusions on population-level risks.

Inherent practices in risk assessment result in exposure estimates that are biased high in order to be reasonably conservative include:

- The use of an upper exposure value (i.e., 95% UCL) to calculate dietary exposure concentrations
- The use of a maximum values in the place of a UCL when limited data are available
- The assumption that a fish species feeds exclusively or is getting all of its seasonal and lifetime exposure to COPECs, from the NBSA (i.e., SUF = 1)

For dietary models and HQs, the assumption that COPECs measured in sediment and/or biota using strong chemical extractants are 100% bioaccessible via dietary uptake (dermal absorption via the gut). This uncertainty is due to the relative bioaccessibility of laboratory feed versus natural diets. When compared to TRVs based on laboratory toxicity tests conducted with recently spiked sediment, exposure doses estimated assuming 100% bioaccessibility are likely to overestimate potential for risk.

As a result, risk estimates are expected to be overestimated rather than underestimated in this BERA, especially with respect to the dietary HQs. Many of these conservative assumptions (e.g., SUF =1, 100% bioaccessibility) are not necessary when HQs are based on directly measured tissue concentrations. While the use of tissue HQs reduces uncertainty in the exposure estimates, there is typically a limited toxicity dataset available with which to derive TRVs. Therefore, uncertainty in the HQs can be biased in either direction (high or low), based on the strength of the dataset used to derive the TRVs.

For risk estimates based on tissue, dietary, and egg HQs, TRVs are a key source of uncertainty in the risk estimates and are discussed below or in Appendix D, as relevant. Additional sources of uncertainty were identified and evaluated either quantitatively or qualitatively. Table 7-18 summarizes the quantitative uncertainty evaluations conducted for fish for the tissue, dietary, and egg assessments and their associated effect on risk estimates. Risk calculations and HQs based on the quantitative uncertainty evaluations are presented in Appendix F and discussed in the following sections. Qualitative evaluations were also conducted and are presented below in their respective sections.

Table 7-18. Summary of Quantitative Fish Uncertainty Evaluations

Species	General Uncertainty Evaluated	Specific Evaluation Conducted	Difference in HQ Values	Effect on Risk Conclusions[a]
Fish Tissue Assessment				
All	Treatment of NDs as zero for EPC calculation	EPCs recalculated using half or full DL	HQs[b] increase (<0.2)	No change
Fish Dietary Assessment				
Summer/winter flounder; White perch	SUF of 0.67	HQs recalculated using SUF of 1	HQs increase (33%)	~~Cadmium~~Flounder: LOAEL HQs based on NBSA TRVs increased to ≥ 1 for methylmercury in N and SE zones, cadmium in N zone, mercury in SE zone; Perch: LOAEL HQs based on NBSA TRVs increased to ≥ 1 for mercury in NBSA-wide, SE, SW zones.
All	Treatment of NDs as zero for EPC calculation	EPCs recalculated using half or full DL	HQs[b] increase (<0.01)	No change
American eel (≥ 50 cm); Summer/winter flounder	Fish prey size of < 30 cm assumed	EPCs recalculated using all prey fish	No change	No change
Fish Egg Assessment				
All	Treatment of NDs as zero for EPC calculation	EPCs recalculated using half or full DL	No change (100% FOD)	No change

Table 7-18 **Notes**:
a. Based on LOAEL HQs ≥ 1.
b. NBSA-wide only results presented in this table. Differences for assessment zones are discussed in the text. All results of the sensitivity analyses are provided in Appendix F.

Additionally, relatively large exposure areas were evaluated in this BERA, including the NBSA-wide area and individual assessment zones. While these areas are appropriate for risk assessment, especially for population-level assessment of mobile fish communities, evaluation of smaller features, such as individual wetlands or mudflats, may be required in the FS. The uncertainty associated with applying the results of the NBSA-wide and assessment zone conclusions to localized areas within the NBSA is considered to be low for highly mobile receptors, and somewhat higher for species with a small relative foraging area, such as mummichog and killifish. For mummichog and killifish, this uncertainty was reduced by evaluating mudflat areas instead of the larger assessment zones in the mummichog dietary assessment, which assumed incidental ingestion of mudflat sediment and ingestion of worm tissue estimated using bioaccumulation testing data collected from mudflat locations.

Another overarching uncertainty for the fish risk assessment is the fact that a formal risk analysis was not conducted for the regularly dredged deep water channels in the NBSA. This was a decision made during the planning of the BERA process due to the changing conditions in these areas from regular navigational dredging activities performed by the USACE, which regularly removes sediments from these channels. Ten composite sediment samples were collected in the regularly dredged deep water channels. Summary

statistics for these samples are presented in Table 4-12. Mean and UCL concentrations for most of the COPECs including 2,3,7,8-TCDD, total PCBs, PAHs, some pesticides, and most metals are generally lower in the channel sediments than those in the other ecological risk assessment zones. Exceptions include total DDx, chlordane, mercury/methylmercury, silver, and zinc for which the average and UCL concentrations are within the range of the other zones. Given that the concentrations of COPECs in the channels are below or within the range of those found in the other zones, the potential for risks to fish receptors that may be exposed in the channels are bounded by the risk estimates for the other zones presented in this BERA.

~~8.6.1~~7.6.1 Fish Community Risk Characterization Uncertainties

As described in Section 7.1.1, the ABS conducted by USACE (2015) generated a robust fish dataset, including survey data collected over eight years for demersal finfish and icthyoplankton (juveniles, larvae, and fish eggs), to evaluate the seasonal distribution and abundance of the fish community in the NY/NJ Harbor Estuary, including the NBSA. ~~This extensive~~

The ABS studies were not designed to evaluate the potential effects of contamination in the NBSA on fish communities. Without comparative long-term data for an appropriate reference estuary or defined statistical power, it is not possible to determine from these historical datasets whether or not COPECs in the Bay are impacting the NBSA fish community~~ survey~~.

These fish community surveys found that fish communities in the NBSA are typical of other mid-Atlantic estuaries, ~~with~~and that generally adult abundance and diversity in the NBSA appear to be similar to or greater than other locations in the NY/NJ Harbor Estuary.~~ These results are considered a strong LOE supporting the health of fish communities in the subtidal zones from which data were collected.~~ The ABS did not focus on intertidal areas, where some fish such as mummichog and killifish are prevalent. Therefore, there is greater uncertainty in applying the conclusion of the ABS to these species that prefer shallow and muddy intertidal habitats.

~~Importantly, the ABS specifically targeted icthyoplankton, including larvae and eggs, and did not find evidence of COPEC-related adverse effects in fish early life stages.~~ Differences in icthyoplankton communities among the regions were attributed by USACE (2015) to physical differences such as bottom substrate, water temperature, salinities, and effects of freshwater inputs that can influence the presence of certain species. Uncertainty associated with the ABS icthyoplankton dataset is greatly reduced relative to the icthyoplankton dataset collected as part of the NBSA investigation, which did not use sampling gear specifically targeted for larvae and eggs. The ABS icthyoplankton dataset provides an independent ~~and strong~~ LOE to help support the conclusions of the fish egg assessment (which has its own associated uncertainties; Section 7.6.3), that adverse effects to early life stages are unlikely in the NBSA. As with any biological community investigation, there is uncertainty in the ability of the population data collected to detect potential effects of stressors to specific species or in localized areas. However, this uncertainty is diminished by the collection of multiple years of data in the same areas, as was the case with the USACE ABS study.

~~8.6.2~~7.6.2 Fish Tissue Risk Characterization Uncertainties

Direct measures of COPECs in samples of fish tissue inherently integrate many factors related to fish COPEC exposure, such as site use and dietary composition, typically resulting in a low degree of uncertainty in the tissue EPCs. By comparison, there is generally greater uncertainty in the tissue effects assumptions and TRVs used to calculated tissue HQs. Toxicity inputs to the tissue HQs, including the use of a dioxin/furan/PCB TEQ methodology and constituent-specific TRV uncertainties, are discussed in Appendix D.

~~8.6.2.1~~7.6.2.1 Toxicology/TRVs

The most important uncertainty associated with the TRVs for the fish tissue assessment is the large difference between the NBSA and LPR FFS TRVs for dioxins/furans and PCBs. The NBSA TRVs for both COPECs were derived by Windward (2019) as 5th percentiles of the SSDs for multiple (mostly freshwater) species, whereas the LPR FFS TRVs were based on single species studies (mummichog and Atlantic salmon for 2,3,7,8-TCDD and PCBs, respectively). The NBSA LOAEL TRVs for these COPECs are higher than the LOAEL TRVs developed for the LPR FFS (USEPA 2014) by approximately two orders of magnitude for 2,3,7,8-TCDD/TEQ and one order of magnitude for total PCBs. Given the inherent uncertainties in the use of reported critical body residues from the literature as TRVs for population-level risk assessments (i.e., due to their limited availability and the unknown applicability in extrapolating from single studies/limited endpoints to population-level risks), it is appropriate to consider a range of TRVs in risk characterization. This bounding approach was taken in the fish risk assessment (and elsewhere in this BERA) to provide perspectives on potential risks, and to maintain consistency with the LPR FFS and LPRSA BERAs. Discussions regarding the specific derivation of these TRVs are provided in Appendix D.

The ~~risk conclusions for the NBSA BERA however, are primarily based on the NBSA TRVs, as these are most representative of the receptors and assessment endpoints (i.e., maintenance of growth, reproduction, and survival of fish populations) established in the Problem Formulation (Section 3). For total PCBs, the NBSA TRV is based on a study of reproduction in mummichog (Black et al. 1998), a resident species in the Bay. The LPR FFS TRV for PCBs is based on a study of behavioral effects (prey capture behavior) in Atlantic salmon (Lemar et al. 2007), a species not represented in the Bay ecosystem, and an endpoint not consistent with the fish assessment endpoint or clearly tied to adverse effects at the individual or population level (specified by USEPA as effects on growth, reproduction, and survival). The LPR FFS LOAEL TRV for PCBs is more than seven times lower than the NBSA LOAEL TRV. While the LPR FFS TRV provides some~~ use of single study TRVs from the LPR FFS as bounding ~~perspective and is evaluated in the HQ calculations, it is~~ estimates for fish risk are more uncertain than the LPRSA BERA SSDs not only because they are single studies, but also because the endpoint of behavior (prey capture) alterations was considered relevant in both 2,3,7,8-TCDD/TEQ and PCB TRVs. The extrapolation of behavioral endpoints to potential population effects are highly uncertain~~ in terms of its relevance to~~. Further, the ~~NBSA fish tissue assessment and is considered to substantially overestimate HQs.~~

~~Similarly,~~ total PCB TRVs are based on Atlantic salmon, which does not occur in the NBSA. The LPR FFS TRV for 2,3,7,8-TCDD~~ and related dioxin/furan and PCB TEQs,~~/TEQ is based on a study ~~(Salomon 1994) on mummichog growth. The LPR FFS TRV is also based on a study of~~ of mummichog (Couillard et al. 2011), which is a resident species in the NBSA, but is ~~a~~an broad extrapolation of apparent behavioral effects in larvae from a single PCB congener (PCB 126), and estimates of larval doses without any

measured tissue concentrations to confirm the larval exposure levels. More importantly, no delayed hatching, growth, malformation, or mortality was found in mummichog in ~~this~~the Couillard et al. (2011) study, even at the highest dose level (50 pg ~~PCB126~~PCB 126 per egg). This study has ~~never~~not been used to derive a 2,3,7,8-TCDD or TEQ TRV for risk assessment in the past. ~~The applicability of deriving a TEQ-based TRV from a single PCB congener is irregular and uncertain. Given these large uncertainties, and the very low TRV (more than 160 times lower than the NBSA TRV, which is based on reduced growth in a resident species) that has been~~ While these studies have a higher degree of uncertainty than the multispecies SSD-derived ~~from this study, the application for the NBSA is limited to discussion~~TRVs, they are utilized for bounding purposes~~. Additionally, both of the sets of tissue TRVs are based on single studies using mummichog as a test species. While mummichog are resident to Newark Bay, and thus directly relevant, they may not be among the most sensitive species marine species. For comparison, the TCDD TRVs for freshwater fish derived in the LPRSA BERA (Windward 2019) are based on an SSD for seven species. The 5th percentile SSD value selected as the LOAEL TRV for the LPRSA BERA is the same as the NOAEL concentration (0.00012 mg/kg ww) reported in Salomon (1994). The NBSA LOAEL TRV of 0.0003 mg/kg ww corresponds with approximately the 12th percentile in the LPRSA BERA SSD and is lower than effect levels included in the LPRSA BERA SSD for sensitive species such as rainbow trout. Thus, use of the NBSA LOAEL TRV, 0.0003 mg/kg ww based on mummichog data (Salomon 1994), is not expected to substantially underestimate toxicity to other marine species. The LPRSA BERA LOAEL TRV and NBSA LOAEL TRV are both approximately two orders of magnitude higher than the LOAEL TRV developed for the LPR FFS (USEPA 2014). In the absence of toxicity data for marine species other than mummichog, it is presumed that TRVs derived for the NBSA are adequately protective of other potentially more sensitive fish species~~ in the fish tissue assessment.

For metals, whole-body tissue concentrations of total metals (other than organometals [i.e., methylmercury, organo-selenium]) are poor predictors of adverse effects for the following reasons: 1) toxicity varies based on the specific metal species present, 2) fish can store excess metals in non-toxic compartments, and 3) toxicity is strongly dependent on the uptake and physiological processing rate and exposure pathway. The use of a tissue residue approach for most metals (except for methylmercury and selenium) is highly uncertain because of the wide range of strategies used by organisms to store, detoxify, and excrete bioaccumulated metals (e.g., fish and invertebrates may regulate their body burdens of some metals, although metals regulation, and the strategy thereof, is species and metal specific) (Adams et al. 2011; USEPA 2007b). For these reasons, USEPA risk assessment guidance for regulated metals and recent expert guidance conclude that comparison of whole-body metals tissue concentrations to literature-reported whole-body effects thresholds is not sufficiently robust for drawing risk conclusions (Adams et al. 2011; USEPA 2007b). For all inorganic COPECs, fish dietary TRVs were used to provide another perspective on the potential for fish toxicity. TRVs for seven of the ten inorganic COPECs were sourced from the LPRSA BERA (Windward 2019), and for the other three (arsenic, lead, and silver) they were developed specifically for the NBSA and based on single studies. However, they still relate back to the LPRSA BERA, as they were studies that were reviewed/evaluated in that report (Windward 2019, Appendix A3-1). Further details on key uncertainties for individual metals are presented in Appendix D.

~~8.6.2.2~~7.6.2.2 Treatment of Non-Detects

The concentrations of summed components (e.g., PCB congeners) that were not detected were assumed to be zero when calculating totals. This assumption may underestimate summed component

concentrations if the individual constituents are actually present at concentrations close to the ~~reporting~~quantitation limit. Therefore, the effect on HQs of using one-half the ~~detection~~quantitation limit or the full ~~detection~~quantitation limit was evaluated quantitatively. As noted in Table 7-18, there were no changes from LOAEL HQ < 1 to HQ ≥ 1 in the fish tissue HQs, regardless of how non-detected values in sums were treated (either as zero, one-half the ~~detection~~quantitation limit, or the full ~~detection~~quantitation limit). Summary tables of HQ results from this sensitivity analysis on the treatment of non-detects, both NBSA-wide and by assessment zones, are ~~found~~presented in Appendix F. The overall effect of using a zero concentration for non-detected constituents when calculating a summed total (e.g., total PCBs, dioxin/furan TEQ, PCB TEQ, PCB/dioxin/furan TEQ) on the HQs is minimal and does not impact risk conclusions for fish.

~~8.6.3~~7.6.3 Fish Dietary Risk Characterization Uncertainties

While dietary exposure modeling has been a well-established practice for wildlife (avian and mammalian) risk assessments (USEPA 1993; Sample et al. 1997) and oral doses are the principal means of contaminant exposure, dietary exposure modeling has not been widely used for evaluating fish risk (USEPA 2007b; Luoma and Rainbow 2005). It is only included in the NBSA BERA to evaluate the potential for risks from metals and PAHs that may not have been adequately captured in the fish tissue assessment. There is substantial uncertainty in both the exposure and effects assumptions used to estimate fish dietary HQs. These uncertainties are discussed below.

~~8.6.3.1~~7.6.3.1 Toxicology/TRVs

Fish dietary TRVs are a fundamental source of uncertainty because most toxicity studies do not report daily doses. Instead, they report adverse effects based on the concentrations in food which then need to be converted to a daily dose with the use of additional variables: body weight (kilograms), ingestion rate (kilograms per day), and food moisture content (percent). Studies typically do not report these variables. In those studies that did report ingestion rate, 2% of the body weight was the most common ingestion rate and this value was used to estimate a daily dose when not reported in laboratory toxicity studies. Moisture content in food was estimated to be 80%, where not reported. Another uncertainty is that food provided in laboratory studies typically contains highly bioavailable forms of the contaminant being tested, which then may partition to the water column. In natural environments, contaminants are in equilibrium and total concentrations in prey, water, and sediment contain both bioavailable and inert fractions of contaminants. Not understanding the degree of contaminant partitioning, bioavailability, and ingestion rates makes this modeling highly uncertain, and it is unclear whether the overall effect is to over- or underestimate fish daily intakes of COPECs.

Metals dietary toxicology is uncertain and has been questioned as a means to quantify dietary risk, specifically with regard to the high variability in uptake and toxicity of inorganic metals in fish. The USEPA has recommended that this dietary approach for fish be used “only for conservatively screening for exposure and potential risks to consumers (i.e., in cases where whole-body residues in prey are below dietary toxic thresholds)” (USEPA 2007b). For more definitive assessments, USEPA suggests that further research is needed to quantify the bioavailability and effects of inorganic dietary metals (USEPA 2007b). A key uncertainty regarding dietary metals is that fish actively regulate the uptake and distribution of essential metals (e.g., copper and zinc) necessary for normal metabolic function and non-essential metals

may also be regulated by these systems because they mimic essential elements (Bury et al. 2003). More recent interest and advances in quantifying dietary trophic transfer of metals in marine systems has been developed by Wang (2002), with a kinetic model, and Luoma and Rainbow (2005) with a biodynamic model. Both models require understanding of variables that may not easily be known (e.g., assimilation efficiency, metal efflux rate). Both Wang (2002) and USEPA (2007b) note a challenge with metals bioaccumulation/bioconcentration estimation because there are over orders of magnitude variation within and between species. Metal concentrations in animals are probably not related to the trophic level in the food web (Wang 2002).

Unlike metals where uptake and toxicity occur through specific mechanisms, PAH uptake in fish occurs through non-specific means with general narcosis, a reversible nervous system impairment, being a noted toxicological effect (Di Toro et al. 2000). Other studies point to more specific mechanistic activity, such as binding to the aryl hydrocarbon receptor (AhR) and CYP1A induction, that can be associated with sublethal effects in early life stages. CYP1A induction is the activation of a detoxification system in fish that allows them to metabolize and eliminate PAHs (Eisler 1987), thus, tissue PAH concentrations may be less correlated to adverse effects. Additionally, recent studies have indicated that mechanisms other than narcosis and AhR-medated toxicity are potentially responsible for PAH toxicity in early life stages of fish. Barron et al. (2004) found that an alkyl phenanthrene toxicity model, rather than narcosis or AhR agonism, best predicted toxicity in early life stages of Pacific herring and pink salmon. Some evidence suggests that alkyl-phenanthrene metabolites are more toxic than the parent compound due to the presence of benzylic alcohols. Incardona et al (2005) found that toxicity of LMW tricyclic PAHs on cardiovascular function and morphogenesis in zebra fish embryos was independent of AhR. As alkylated PAHs are the predominant form of PAHs in oil, evaluation of only non-alkylated PAHs, as was done in this BERA, may underestimate exposure and risk associated with PAHs in the NBSA. The magnitude of this uncertainty is unknown. However, the calculated HQs for total PAHs are orders of magnitude below 1 and additional LOEs evaluated in this BERA, including fish pathology and community surveys, are consistent with the HQs. Although potential for toxicity associated with akylated PAHs was not included in this BERA, based on consideration of other factors discussed above, risk conclusions for fish and PAHs in the NBSA are not expected to change.

~~8.6.3.2~~7.6.3.2 Quantitative Exposure Uncertainty Assessment

Several uncertainties were quantitatively evaluated to assess potential impact on the risk characterization. These uncertainties are presented below and summarized in Table 7-18.

~~8.6.3.2.1~~7.6.3.2.1 Site Use Factor

Based on data from fish surveys in the NBSA and the literature, summer/winter flounder and white perch receptors SUF were assumed to be 0.67 in this BERA (all other fish diet receptors species had SUFs = 1). A SUF of 1 was conservatively evaluated in this sensitivity analysis. Summary tables of HQ results from this sensitivity analysis on the effect of SUF values are presented in Appendix F. When the dietary food web models were analyzed using a SUF = 1 rather than 0.67 for summer/winter flounder, the cadmium and methylmercury north and southeast zone LOAEL HQs based on NBSA TRVs changed from < 1 to ≥ 1.When the dietary food web models were analyzed using a SUF = 1 rather than 0.67 for white

perch, the NBSA-wide, Southeast, and Southwest Zone mercury LOAEL HQs based on NBSA TRVs changed from < 1 to ≥ 1.

~~8.6.3.2.2~~7.6.3.2.2 *Treatment of Non-Detects*

For calculation of EPCs, the concentrations of summed components (e.g., total PAH) that were not detected were assumed to be zero when calculating totals. The effect on HQs of using one-half the ~~detection~~quantitation limit or the full ~~detection~~quantitation limit was evaluated. There were no changes from LOAEL HQ < 1 to HQ ≥ 1 for NBSA-wide or zone HQs regardless of how non-detected values in sums were treated (either as zero, one-half the ~~detection~~quantitation limit, or the full ~~detection~~quantitation limit). Summary tables of HQ results from this sensitivity analysis on the treatment of non-detects are presented in Appendix F. The overall effect of using a zero concentration for non-detected constituents when calculating a summed total on the HQs is minimal and does not impact risk conclusions for fish.

~~8.6.3.2.3~~7.6.3.2.3 *Fish Prey Sizes*

Based on data from the literature, only prey fish < 30 cm were included in the fish dietary components for summer/winter flounder (30% of diet included prey fish < 30 cm) and American eel ≥ 50 cm (40% of diet included prey fish < 30 cm). This diet excluded several larger fish samples, which may contain higher concentrations of some bioaccumulative COPECs. Summer/winter flounder and American eel ≥ 50 cm dietary food web models were analyzed using all fish samples to assess this uncertainty. There were no changes to fish dietary LOAEL HQs when all fish samples were incorporated into the dietary assessment. The calculated EPCs for fish of all lengths used and the summary tables of HQ results from this sensitivity analysis on the effect of including fish of all lengths in the fish dietary risk evaluation are found in Appendix F. The overall effect of using only prey fish < 30 cm on the HQs is minimal and does not impact risk conclusions for fish.

~~8.6.3.3~~7.6.3.3 Other Uncertainties

The following additional uncertainties associated with the dietary assessment were identified but not quantitatively evaluated:

- Modeled diet – Dietary items were limited to prey species with tissue chemistry data available from the NBSA, including bioaccumulation worm, blue crab, softshell clam, and fish tissue, and did not include other prey items that may be important components of fish diets, such as amphipods, copepods, algae, zooplankton, squid, shrimp, or detritus. These unsampled prey types often have lower tissue concentrations of many COPECs in relation to sampled prey types (crabs, bivalves, and fish) that feed higher on the food change and/or effectively sequester some COPECs (i.e., some metals). The exclusion of these various unsampled prey types likely results in overestimation of exposure and risk for predator fish in this BERA. In addition to the limited types of prey used in the fish dietary model, the selection of explicit prey portions did not reflect the largely opportunistic feeding behavior of most fish species. Therefore, the representativeness of the estimated dietary compositions (based on available prey tissue data) for actual NBSA fish diets is uncertain.
- Food ingestion rate – Food ingestion rates for all fish were estimated as a function of body weight and water temperature using an equation from Arnot and Gobas (2004). It is unknown whether the

modeled food ingestion rates reflect actual ingestion rates of NBSA fish or whether higher or lower ingestion rates could substantially impact risk estimates.

- Sediment ingestion rate - Measured incidental sediment ingestion rates for all fish species other than American eel were not available. Sediment ingestion rates for these species were based on what is known about each species' feeding habitats and best professional judgment. It is unknown whether the estimated sediment ingestion rates reflect actual ingestion rates of NBSA fish or whether higher or lower ingestion rates could substantially impact risk estimates.
- Representativeness of worm dietary fraction – Dietary compositions evaluated for all fish receptors contained some fraction of worm tissue, which was estimated in the BERA based on concentrations measured in polychaete bioaccumulation tests. The bioaccumulation tests were conducted with field sediments collected from eight locations in subtidal and intertidal mudflats along the shoreline of the NBSA (Figure 4-2). These 8 locations represent a variety of conditions, including near-shore historically disturbed areas, industrial waterfront areas, areas designated as shoreline habitat, and open subtidal mudflats further offshore. Although a relatively small number of locations were tested, the variety of conditions makes it more likely to represent the range of worm concentrations that fish and wildlife might consume as they forage throughout the NBSA. With respect to the variety of species consumed by fish and wildlife, the use of worm tissue EPCs generated from laboratory bioaccumulation tests likely overestimates invertebrate dietary exposure due to the high uptake rates for worms relative to other invertebrate prey species, as well as the potential for increased bioavailability of contaminants in sediments that occurs during sample processing for bioaccumulation testing in the laboratory.

~~8.6.4~~7.6.4 Fish Egg Tissue Risk Characterization Uncertainties

This section discusses the uncertainties associated with exposure assumptions, EPCs, and selected TRVs that could affect HQ calculations for fish eggs. Uncertainties associated with the TEQ methodology and general TRV uncertainties are discussed in Appendix D.

Fish egg TRVs were used in the assessment of 2,3,7,8-TCDD/TEQ risk. These TRVs were the same for the LPRSA BERA (Windward 2019) and LPR FFS (USEPA 2014) and were based on the SSD of 10 species derived by Steevens et al. (2005). This consensus suggests more certainty in this endpoint TRV. For total PCBs, there were TRVs for fish egg tissue, but only from the LPRSA BERA (Windward 2019) and the basis of these TRVs was a single study making them less certain than TRVs based on an SSD.

Consistent with the LPRSA BERA, the fish egg assessment was only conducted for resident *Fundulus* sp. that are known to spawn in the NBSA, and for a limited group of bioaccumulative COPECs. The egg assessment was conducted to provide additional risk perspective to the adult fish tissue assessment. One key uncertainty is that adults of other species, particularly the white perch, had higher concentrations of these COPECs in their tissues. For this reason, it is possible that such species may have higher concentrations of COPECs in their eggs as well. However, these fish are migratory and are exposed to COPECs throughout their range, and do not spawn in the Bay. The risk to these fish is assessed from the adult tissue and dietary exposures; however, there is a possibility that their exposure times and COPEC accumulation from the NBSA may cause adult:egg transfers of COPECs that are a risk.

The greatest uncertainty in this assessment is the use of an adult-to-egg uptake factors from the literature to estimate mummichog egg concentrations.

- For organics (i.e., PCBs and TEQs), a predictive relationship exists between adult whole-body tissue and egg tissue, and the resulting regression equation was used to estimate egg concentrations. Although the regression is based on empirical data, it is unknown whether the literature-based data over- or under-predicts NBSA mummichog egg concentrations (and risks) due to differences in exposure conditions, bioavailability of PCBs, and different uptake characteristics of the specific PCB mixtures tested. The uncertainty associated with modeled egg concentrations can be evaluated by considering the range of CFs evaluated (0.62 – 1). The use of CFs based on Niimi (1983) vs. Russell et al. (1999) slightly changes the HQs for PCBs and TEQ - fish COPECs; although, it does not change whether or not a HQ is above or below 1. While theoretically possible for maternal transfer to result in a CF greater than 1, the approach and range of CFs used in this BERA is consistent with that approved for the LPRSA BERA (Windward 2019), and therefore the magnitude of these uncertainties are considered acceptable. Additionally, the CF assumes a lipid content of 3.3% in mummichog eggs. This value is based on the 95UCL measured in mummichog eggs in the LPRSA (Windward 2019). While appropriate for evaluation of mummichog, it may result in over- or underestimation of egg concentration in different species, depending on their egg lipid content. The relevance of the egg HQs for other species is tied to the similarity in egg lipid contents to mummichog. The assumed lipid content of 3.3% for the fish egg analysis is within the range of the lipid content 95UCLs of other fish species sampled in Newark Bay BERA sampling program, including American eel (7.1%), Atlantic silverside (1.6%), Atlantic menhaden (5.1%), summer flounder (1.2%), white perch (3.1%), and winter flounder (1.3%).
- For methylmercury, there was no relationship between adult and egg tissue concentrations in the literature data, and therefore a simple bioconcentration factor was selected based on the available information. The use of a bioconcentration factor to estimate methylmercury concentrations in eggs is uncertain; however, the use of the maximum CF of 0.101 likely over-predicts egg tissue concentrations (and HQs) for NBSA fish. The range of available fish CFs for methylmercury is 0.039 to 0.101 (Niimi 1983), based on data for five species (rainbow trout, smallmouth bass, white bass, white sucker, and yellow perch). Based on these data, the maximum CF and CF of 1 evaluated in this BERA may also over-predict potential risks to mummichog.

As summarized in Table 7-18, the following additional uncertainties were quantitatively evaluated:

- Treatment of non-detects for EPCs - The concentrations of summed components (e.g., PCB congeners) that were not detected were assumed to be zero when calculating totals. The effect on HQs of using one-half the ~~detection~~quantitation limit or the full ~~detection~~quantitation limit was evaluated. There were no changes from LOAEL HQ < 1 to HQ ≥ 1 for any COPEC regardless of how non-detected values in sums were treated (either as zero, one-half the ~~detection~~quantitation limit, or the full ~~detection~~quantitation limit).

~~8.7~~7.7 Weight of Evidence Risk Findings for Fish

As summarized in Section 3.5, a WOE approach was used to evaluate COPECs with LOAEL HQs ≥ 1 (preliminary COCs; Table 7-19) and potential for unacceptable risk to fish populations in the NBSA.

Table 7-19. Summary of Preliminary COCs for Fish Tissue, Dietary, and Egg Assessments

Preliminary COCs	Exposure Area	Range of LOAEL HQs														
		Fish Tissue LOE								Fish Egg LOE[1]		Fish Diet LOE				
		NBSA TRVs				LPR FFS TRVs				NBSA TRVs	LPR FFS TRVs	NBSA TRVs				
		Mummichog	American Eel	Flounder	White Perch	Mummichog	American Eel	Flounder	White Perch	Mummichog		Mummichog	Flounder	White Perch	American Eel < 50 cm	American Eel ≥ 50 cm
2,3,7,8-TCDD	NBSA-wide	<1	<1	<1	<1	14	4	4	39	<1	<1	--	--	--	--	--
	Zones	<1	<1	<1	<1 - 1	9 - 21	4 - 8	2 - 7	16 - 69	--	--	--	--	--	--	--
Total Dioxin/Furan TEQ Fish	NBSA-wide	<1	<1	<1	<1	16	6	4	40	<1	<1	--	--	--	--	--
	Zones	<1	<1	<1	<1 - 1	10 - 23	6 - 8	2 - 7	20 - 73	--	--	--	--	--	--	--
Total Dioxin/Furan TEQ Fish Alternative TRV	NBSA-wide	1	<1	<1	3	NC	NC	NC	NC	--	--	--	--	--	--	--
	Zones	<1 - 2	<1	<1	2 - 6	NC	NC	NC	NC	--	--	--	--	--	--	--
Total PCBs	NBSA-wide	<1	<1	<1	<1	3	1	<1	3	3 - 5	NC	--	--	--	--	--
	Zones	<1	<1	<1	<1	2 - 6	1 - 2	<1 - 1	3 - 4	--	--	--	--	--	--	--
Total PCB TEQ Fish	NBSA-wide	<1	<1	<1	<1	1	<1	<1	<1	<1	<1	--	--	--	--	--
	Zones	<1	<1	<1	<1	<1 - 3	<1 - 1	<1	<1 - 1	--	--	--	--	--	--	--
Total Dioxin/Furan/PCB TEQ Fish	NBSA-wide	<1	<1	<1	<1	17	6	4	40	<1	<1	--	--	--	--	--
	Zones	<1	<1	<1	<1 - 1	11 - 24	6 - 9	2 - 8	21 - 90	--	--	--	--	--	--	--
Total Dioxin/Furan/PCB Fish Alternative TRV	NBSA-wide	1	<1	<1	3	NC	NC	NC	NC	--	--	--	--	--	--	--
	Zones	<1 - 2	<1	<1	2 - 7	NC	NC	NC	NC	--	--	--	--	--	--	--
Total DDx (2,4 & 4,4)	NBSA-wide	<1	<1	<1	1	<1	<1	<1	1	<1	<1	--	--	--	--	--
	Zones	<1	<1	<1	<1 - 2	<1 - 1	<1 - 1	<1	1 - 2	--	--	--	--	--	--	--
Arsenic[2]	NBSA-wide	<1	<1	<1	<1	NC	NC	NC	NC	--	--	<1	<1	<1	<1	<1
	Zones	<1	<1	<1	<1	NC	NC	NC	NC	--	--	<1	<1	<1	<1	<1
Cadmium	NBSA-wide	--	--	--	--	--	--	--	--	--	--	1	1	1	1	1
	Zones	--	--	--	--	--	--	--	--	--	--	1 - 2	<1 - 1	1 - 1	1 - 2	1 - 1
Chromium[2]	NBSA-wide	<1	<1	<1	<1	NC	NC	NC	NC	--	--	NC	NC	NC	NC	NC
	Zones	<1	<1	<1	<1 - 1	NC	NC	NC	NC	--	--	NC	NC	NC	NC	NC
Copper[2]	NBSA-wide	1	<1	<1	5	3	<1	<1	14	--	--	<1	<1	<1	<1	<1
	Zones	<1 - 2	<1	<1	3 - 8	2 - 6	<1 - 2	<1 - 1	8 - 23	--	--	<1	<1	<1	<1	<1
Mercury	NBSA-wide	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	1	<1	1	1
	Zones	<1	<1	<1	<1	<1	<1 - 1	<1	<1	--	--	<1 - 1	<1 - 1	<1 - 1	1 - 2	1 - 2
Methylmercury	NBSA-wide	<1	<1	<1	<1	<1	<1	<1	<1	<1 - 1	<1 - 1	<1	<1	<1	<1	1
	Zones	<1	<1 - 1	<1	<1	<1	<1 - 1	<1	<1	--	--	<1	<1	<1	<1 - 1	1
Selenium[2]	NBSA-wide	<1	<1	<1	<1	NC	NC	NC	NC	--	--	<1	<1	<1	<1	<1
	Zones	<1	<1 - 1	<1	<1 - 1	NC	NC	NC	NC	--	--	<1	<1	<1	<1	<1
Silver[2]	NBSA-wide	<1	<1	<1	2	NC	NC	NC	NC	--	--	NC	NC	NC	NC	NC
	Zones	<1 - 1	<1	<1	1 - 3	NC	NC	NC	NC	--	--	NC	NC	NC	NC	NC
Zinc[2]	NBSA-wide	<1	<1	<1	<1	NC	NC	NC	NC	--	--	<1	<1	<1	<1	<1
	Zones	<1	<1	<1	<1	NC	NC	NC	NC	--	--	<1	<1	<1	<1	<1

Notes:

[1] HQ range includes two different conversion factors (see Table 7-15 for more details)

[2] White perch liver LOAEL HQs ≥ 1 for the following inorganics and assessment zones (see Table 7-9 and Section 7.1.3.2 for more details): Arsenic in NBSA-wide and SW zone; Chromium in NBSA-wide and SW zone; Copper in NBSA-wide, N, SE, SW zones; Selenium in NBSA-wide, N, SE, SW zones; Silver in NBSA-wide, N, SE, SW zones; Zinc in NBSA-wide, N, SE, SW zones

NC = not calculated; TRV not available

-- = not evaluated

For fish, long-term community monitoring data collected under the ABS (USACE 2015a, 2015b), a range of dietary and tissue HQs~~, and~~ (including sensitivity analyses), a semi-quantitative evaluation of fish pathology, site-specific toxicity data (for killifish; Bugel et al. 2010), and consideration of background sediment concentrations (Section 4.4.2) provide LOEs used in this BERA. The range of HQs includes:

1. Tissue HQs based on whole-body or liver tissue (white perch only) concentrations measured in fish compared to tissue TRVs derived for the NBSA.
2. Tissue HQs based on whole-body and liver tissue concentrations (white perch only) measured in fish collected in the NBSA compared to tissue TRVs derived for the LPR FFS (USEPA 2014).
3. Dietary HQs based on dietary doses in food web models estimated using measured prey tissue concentrations compared to literature-derived dietary TRVs specific to the NBSA.
4. Egg tissue HQs based on modeled egg concentrations in mummichog/killifish compared to egg TRVs derived for the LPR FFS (USEPA 2014).

Tissue HQs were calculated for the NBSA-wide area and also independently for the north, southeast, and southwest assessment zones where sufficient tissue data were available. Dietary HQs were calculated for mummichog/killifish, American eel, flounder (winter and summer combined), and white perch for the NBSA (mudflats only for mummichog) and individual assessment zones. Egg tissue HQs were estimated for bioaccumulative compounds for the NBSA-wide area.

For fish community data, the ABS (USACE 2015a, 2015b) was a long-term, diverse and spatially robust sampling program that included bottom trawl surveys of demersal or near bottom occurring finfish (2002 – 2010 dataset) and ichthyoplankton tows using a benthic sled equipped with a plankton net to collect fish eggs and larvae near the bottom (2002 – 2011 dataset). Sampling locations included Newark Bay and Arthur Kill, and other Upper and Lower New York Bay locations. Observations of gross physical abnormalities were used as an indicator of site-specific toxicity as well.

Uncertainties (Section 7.6) were evaluated in this WOE assessment according to the factors presented in Section 3.5 to arrive at an overall risk conclusion for each preliminary COC. ~~Fish~~As discussed in Sections 7.6.2.1 and 7.6.3.1, fish TRVs are a key uncertainty in the risk estimates and apply to all fish receptors evaluated in this BERA. ~~For preliminary COCs, Table 7-19 indicates which TRVs have highest reliability for risk estimation in the NBSA BERA based on the uncertainties presented in Section 7.6 for whole-body tissue, dietary, and egg TRVs. Fish TRV reliability was considered in the WOE evaluation presented in this section. There are two primary factors affecting reliability of the TRVs as previously mentioned in Section 3.5: 1) The existence of a dose–response relationship and the strength of the relationship increase confidence and reliability in the TRVs and 2) TRVs derived for species present at the site or derived using realistic exposure routes are considered more reliable. The strongest evidence for a dose–response relationship is when an SSD was developed and since the LOAEL TRV developed from an SSD represents the 5th percentile of this distribution there is very good reliability that risk to potential receptors has not been underestimated. If data were not sufficient to develop an SSD and, therefore, the TRV was based on the selection of a single study, the reliability of these TRVs was higher when multiple studies were found acceptable for TRV derivation in which case the selected TRVs were from the study that determined the lowest LOAEL. The TRVs that were considered least reliable were based on a single study without corroboration from other sources. Overall, there was more confidence in the risk estimates when multiple TRVs sourced from the regional risk assessments that have been completed (Windward 2019 and USEPA 2014) could be included in the evaluation as another line of evidence to bound risk estimates. In general, the parties concur that the LPR FFS TRVs (USEPA 2014) are more conservative than comparable TRVs developed by the CPG for the LPRSA BERA (Windward 2019).~~

Table 7-19. Reliability of Risk Estimates for Preliminary COCs Based on Available TRVs

Preliminary COCs	Reliability for Risk Estimation in the NBSA BERA				
	Tissue TRVs		Dietary TRVs		Egg TRVs
	NBSA TRVs	LPR FFS TRVs	NBSA TRVs	LPR FFS TRVs	NBSA & LPR FFS TRVs
Dioxins/Furans & TEQ[a]	**Moderate**; Mummichog TRV, Lowest marine LOAEL	**Low**; Behavioral endpoint, egg exposure to single PCB congener	-	-	**Moderate**; 5%ile SSD with conservative dataset
Total PCBs	**Moderate**; Mummichog TRV; Lowest marine LOAEL; equivalent to 5%ile SSD from LPRSA BERA[d]	**Low**; Behavioral endpoint for salmon	-	-	**Moderate**; Lowest LOAEL
Total DDx	**Moderate**; Lowest LOAEL; Similar to LPRSA BERA 5%ile SSD	**Moderate**; 5%ile SSD with conservative dataset	-	-	-
Arsenic	**Moderate**; Lowest LOAEL[d]	-	**Moderate**; Lowest LOAEL	-	-
Cadmium	-	-	**Moderate**; Lowest LOAEL[b,d]	-	-
Chromium	**Low**; No dose-response relationship in tissue	-	**Low**; Unbounded NOAEL only	-	-
Copper	**Low**; No dose-response relationship in tissue; below background tissue concentrations	**Low**; TRV below nutritional threshold; below background tissue concentrations	**Moderate**; Lowest LOAEL[d]	-	-
Mercury & Methylmercury	**High**; Lowest LOAEL;Similar to LPRSA BERA 5%ile SSD	**High**; 5%ile SSD	**Moderate**; Lowest LOAEL	-	**Low**; Acute endpoint, limited dataset
Selenium	**Moderate**; Similar to LPRSA BERA TRV based on EC10 for most sensitive species	-	**High**; Lowest LOAEL; Similar to LPRSA BERA 5%ile SSD	-	-
Silver	**Low**; Only LOAEL, no dose-response in tissue [d]	-	**Low**; Unbounded NOAEL only	-	-
Zinc	**Moderate**; Lowest LOAEL[c,d]	-	**Low**; Only LOAEL[d]	-	-

For most dietary and tissue TRVs, two sets of TRVs (i.e., NBSA and LPR FFS TRVs) were included for bounding purposes. For risk findings, the two sets of TRVs were given equal weight in this WOE evaluation. Additional LOE are discussed below for each receptor and preliminary COC. **Notes:**

a. TEQ related COPECs include 2,3,7,8-TCDD, Total Dioxin TEQ Fish, Total PCB TEQ Fish, Total PCB Dioxin TEQ Fish

b. This LOAEL is orders of magnitude lower than other available toxicity data for cadmium.

c. Lowest LOAEL above nutritional threshold.

~~d. This value also selected in the LPRSA BERA (Windward 2019).~~
~~- = not applicable; SSD = species sensitivity distribution~~

The risk conclusions based on the WOE are discussed below and summarized by species in Tables 7-20 through 7-23, and the overall summary of fish risks is presented in Table 7-24.

~~8.7.1~~7.7.1 Localized Resident Forage Fish (Combined Mummichog and Killifish)

For mummichog and killifish, whole-body tissue HQs, dietary HQs, and egg tissue HQs were the available HQ LOEs. Fish community surveys (USACE 2015a, 2015b), fish pathology, ~~and~~ a field investigation of killifish reproductive capacity (Bugel et al. ~~2010~~2010), and background sediment concentrations (Section 4.4.2) were also considered in the WOE. LOAEL HQs ≥ 1 are summarized for mummichog/killifish in Table 7-20. Sensitivity analyses included evaluation of NDs for tissue HQs.

Table 7-20. Summary of LOAEL HQs ≥ 1 for Mummichog/Killifish

Preliminary COCs	Tissue LOAEL HQs ≥ 1		Dietary LOAEL HQs ≥ 1		Egg LOAEL HQs ≥ 1	
	NBSA TRVs	LPR FFS TRVs	NBSA TRVs	LPR FFS TRVs	NBSA TRVs	LPR FFS TRVs
Primary Exposure and Toxicity Assumptions (Tables 7-8, 7-13, and 7-16)						
Dioxins/Furans & TEQ (2,3,7,8-TCDD, Total Dioxin TEQ Fish, Total Dioxin/Furan/PCB Dioxin TEQ Fish)	NBSA-wide, N zone~~None~~	NBSA-wide, N, SE, SW zones	-	-	None	None
Total PCBs	None	NBSA-wide, N, SE, SW zones	-	-	NBSA-wide~~None~~	None
Total PCB TEQ	None	NBSA-wide, SW zone	-	-	None	None
Total DDx	None	SW zone	-	-	-	-
Cadmium	-ND	-ND	NBSA-wide, N, SE, SW zones	-	-	-
Copper	NBSA-wide, N zone	NBSA-wide, N, SE, SW zones	None	-	-	-
Mercury	None	None	SW zone	-	None	None
Methylmercury	None	None	None	-	NBSA-wide	NBSA-wide
Silver*	N zone	-	None[a]	-	-	-
Sensitivity Analyses (Table 7-18)						
No additional COCs were identified from sensitivity analyses for this receptor						

Notes:
- = not calculated; ND – not detected
a. The NOAEL HQ was < 1; a LOAEL HQ was not available.
NBSA-wide includes exceedances in each of the three zones unless otherwise noted.
*Sediment COPEC concentrations are within background levels and tissue residues are assumed to be related to background conditions as well.

Tissue (whole-body) HQs were estimated for COPECs with sufficient toxicity data to derive TRVs. Dietary models were used to evaluate COPECs for which tissue concentrations may not be useful for evaluation of toxicity, such as some actively-regulated metals and organics subject to metabolism. Egg tissue HQs were estimated for bioaccumulative COPECs (methylmercury, PCBs, dioxins/furans, and TEQ).

The long-term fish community surveys (USEPA 2015) found that the fish community in the NBSA is comparable to that of other areas in the NY/NJ Harbor Estuary in terms of the species assemblage, life stages, diversity, and seasonal abundance. In addition, it is comparable to what is expected for fish communities in mid-Atlantic estuaries.

The forage fish pathology evaluation found a low percentage of minor pathologies, and does not indicate unhealthy forage fish populations.

A literature search revealed a site-specific study on fish health in the NBSA. Bugel et al. (2010) conducted a field study comparing the reproductive capacity of mummichog in Newark Bay to those from a reference location in the Great Bay, Tuckerton, New Jersey. Several reproductive metrics were

evaluated, including male and female gonad histology; vitellogenin production; and bile chemistry for specific low-, medium-, and high molecular weight PAHs. Examinations of male and female gonad histology show decreased gonadal weight, altered testis morphology in males, and altered gonad development in females. Altered female gonad development was indicated in female Newark Bay mummichog by somewhat higher pre-vitellogenic follicles observed in Newark Bay as compared to Great Bay (64% versus 43%, respectively), lower mid-vitellogenic follicles (17% versus 22%, respectively), and lower mature stage follicles (3% versus 25%, respectively). Overall, ~~Bugel et al. (2010)~~Bugel et al. (2010) concluded that Newark Bay mummichog displayed signs of endocrine disruption and decreased reproductive capacity, despite a lack of significant differences in body size or weight. The specific causes and implications of these histological and biomarker effects on overall reproductive success were not demonstrated by this study and remain unknown.

Tissue HQs inherently integrate many factors related to fish COPEC exposure, such as site use and dietary composition, typically resulting in a low degree of uncertainty in the tissue EPCs. Toxicity inputs to the tissue HQs, such as the constituent-specific TRVs are discussed above as part of the WOE evaluation for fish. For dietary HQs, additional inputs to the HQs, including SUFs and EPCs, are also discussed as part of the WOE.

SUF: Mummichog and killifish are common within the NBSA and were evaluated for exposure to the mudflat areas in the dietary models. The selected SUF of 1 is appropriate based on the small home range (on the order of hundreds of meters or less; Appendix B) for these fish and is not likely to overestimate exposure in either the NBSA-wide area or individual assessment zones.

Diet: In this BERA, mummichog were assumed to have a diet of 100% polychaete worm, but this opportunistic feeder will consume a wide variety of small aquatic invertebrates, algae, and detritus. ~~3~~ The assumed diet likely overestimates exposure to this receptor in the dietary food web models.

EPCs: In general, EPCs were based on 95% UCL concentrations which are appropriate exposure estimates for a BERA. Exceptions include:

- Dioxin/furan TEQ and total PCBs in egg tissue: Egg-tissue EPCs were calculated using adult-to-egg CFs based on a significant relationship between lipid-normalized adult and egg organic COPEC concentrations. A CF of 1.0 was also evaluated as part of a sensitivity analysis, but is not supported by available data for these COPECs.
- Methylmercury in worm prey tissue: Methylmercury was detected in only one of eight polychaete worm samples measured as part of the bioaccumulation investigation. The methylmercury EPC for this prey tissue is based on the maximum detected concentration.
- ~~Methylmercury~~Mercury and methylmercury in egg tissue: For mercury and methylmercury, egg tissue EPCs were calculated using adult-to-egg CFs. Because a significant relationship was not found between adult whole-body and egg ~~methylmercury~~ concentrations, a CF of 0.1 was evaluated based on the maximum CF measured in fish.

~~**Conclusions**:~~**Conclusions**: For most preliminary COCs identified for this receptor, whole-body tissue HQs were considered to have the least uncertainty for drawing risk conclusions. The dietary models have additional uncertainties related to the use of exposure assumptions (SUFs, diet, bioaccumulation factors [BAFs]) and the egg HQs were considered less certain due to the use of a CF to estimate egg

concentrations. For copper and silver, dietary HQs were considered to be the least uncertain, due to the high degree of uncertainty in the tissue TRVs. Based on the various factors considered in the available LOEs, the resulting risk conclusions for preliminary COCs are as follows:

- Dioxins/furans and TEQs: Unacceptable risk is considered ~~unlikely~~possible. These conclusions are based on ~~1)~~ tissue LOAEL HQs ~~< 1~~greater than 10 based on ~~NBSA~~the LPR FFS TRVs~~; 2) egg~~ and tissue LOAEL HQs ~~< 1 even using a conservative CF =1; and 3) fish pathology does not~~greater than 1 based on the alternative NBSA TRV. Other available LOEs indicate ~~unhealthy~~that unacceptable risk is unlikely for forage fish populations.
- ~~Total PCBs: Unacceptable risk is considered unlikely.~~ These ~~conclusions are based on~~include 1) tissue LOAEL HQs < 1 based on NBSA TRVs; 2) egg LOAEL HQs < 1 even using a conservative CF =1; and 3) fish pathology does not indicate unhealthy forage fish populations.
- Total PCB TEQ: Unacceptable risk is considered unlikely. These conclusions are based on 1) tissue LOAEL HQs < 1 using NBSA TRVs; 2) low magnitude of the LOAEL HQs ≥ 1; 3) egg LOAEL HQs < 1 even using a conservative CF =1; and 4) fish pathology does not indicates unhealthy forage fish populations.
- Total PCBs: Unacceptable risk is considered possible in the Southwest Zone. These conclusions are based on 1) tissue LOAEL HQs < 1 based on NBSA TRVs; 2) low magnitude of the tissue LOAEL HQs ≥ 1 using LPR FFS TRVs, except in the Southwest Zone where the HQ = 6; 3) egg LOAEL HQs < 1 even using a conservative CF =1; and 4) fish pathology does not indicate unhealthy forage fish populations.
- Total DDx: Unacceptable risk is considered unlikely. These conclusions are based on 1) tissue LOAEL HQs < 1 using NBSA TRVs; 2) low magnitude of the LOAEL HQs ≥ 1; 3) limited spatial extent (LOAEL HQs ≥ 1 for Southwest Zone only); and 4) fish pathology does not indicates unhealthy forage fish populations.
- Cadmium: Unacceptable risk is considered unlikely. These conclusions are based on 1) cadmium was never detected in whole-body tissue; 2) low magnitude of the LOAEL HQs ≥ 1 based on dietary models; 3) conservative exposure assumptions in the dietary models (100% worm diet); and 4) fish pathology does not indicate unhealthy forage fish populations.
- Copper: Unacceptable risk is considered ~~uncertain, but~~ unlikely. These conclusions are based on conflicting conclusions based on two uncertain HQs methods (dietary and tissue HQs). It should be noted that the dietary HQs incorporate conservative assumptions (100% worm diet), and fish pathology does not indicate unhealthy forage fish populations. In addition, as reported in the LPRSA BERA (Windward 2019), the background concentration of copper in mummichog collected from Jamaica Bay by NOAA (2013) was 4,200 µg/kg. The NBSA forage fish concentrations are at or below this concentration. As such, risks to forage fish from copper are unlikely.
- ~~Silver~~Mercury: Unacceptable risk is considered unlikely. These conclusions are based on 1) whole-body tissue ~~and~~ LOAEL HQs < 1 using both NBSA and LPR FFS TRVs; 2) dietary LOAEL HQs < 1 ~~in the NBSA-wide area; 2)~~ for most scenarios; 3) low magnitude of the dietary LOAEL HQs, equal to 1 in Southwest Zone only; 4) egg LOAEL HQs < 1 even using a conservative CF =1; and 5) fish pathology does not indicate unhealthy forage fish populations.

- Methylmercury: Unacceptable risk is considered unlikely. These conclusions are based on 1) whole-body tissue LOAEL HQs < 1 using both NBSA and LPR FFS TRVs; 2) dietary LOAEL HQs < 1 for all scenarios; 4) egg LOAEL HQs = 1 using a conservative CF =1; and 5) fish pathology does not indicate unhealthy forage fish populations.
- Silver: Unacceptable risk is considered unlikely. These conclusions are based on 1) tissue and dietary LOAEL HQs < 1 in most scenarios, except for tissue HQs in the North Zone; 2) low magnitude of the tissue LOAEL HQ ≥ 1; (North Zone only); and 3) fish pathology does not indicate unhealthy forage fish populations. Although the whole-body tissue and dietary silver TRVs are uncertain, the risk conclusions are supported by an additional LOE, in that sediment concentrations in the NBSA are at or below background concentrations.

8.7.27.7.2 American Eel

For American eel, whole-body tissue HQs and dietary HQs (evaluated for eels <50 cm and ≥ 50 cm) were the available HQ LOEs. Fish community surveys (USACE 2015a, 2015b) and fish pathology were considered as indicators of overall fish population health. Background sediment concentrations (Section 4.4.2) were also considered as a LOE. LOAEL HQs ≥ 1 were predicted for the COPECs listed in Table 7-21. Sensitivity analyses included evaluation of NDs for tissue and dietary HQs, and modified prey size assumptions for the dietary HQs.

Table 7-21. Summary of LOAEL HQs ≥ 1 for American Eel

Preliminary COCs	Tissue LOAEL HQs ≥ 1		Dietary LOAEL HQs ≥ 1 <50 cm		Dietary LOAEL HQs ≥ 1 ≥50 cm	
	NBSA TRVs	LPR FFS TRVs	NBSA TRVs	LPR FFS TRVs	NBSA TRVs	LPR FFS TRVs
Primary Exposure and Toxicity Assumptions (Tables 7-8 and 7-13)						
Dioxins/Furans (2,3,7,8-TCDD, Total Dioxin TEQ Fish, Total Dioxin/ PCB TEQ Fish)	None	NBSA-wide, N, SE, SW zones	-	-	-	-
Total PCBs	None	NBSA-wide, N, SE, SW zones	-	-	-	-
Total PCB TEQ	None	N zone	-	-	-	-
Total DDx	None	SW zone	-	-	-None	-
Cadmium	-ND	-ND	NBSA-wide, N, SE, SW zones	-	NBSA-wide, N, SE, SW zones	-
Copper	None	SW zone	None	-	None	-

Mercury	None	N zone	NBSA-wide, N, SE, SW zones~~None~~	-	NBSA-wide, N, SE, SW zones~~None~~	-
Methylmercury	N zone~~None~~	N, SW zones	SE zone~~None~~	-	NBSA-wide, N, SE, SW zones~~None~~	-
Selenium*	SE zone	-	None	-	None	-
Sensitivity Analyses (Table 7-18)						
No additional COCs were identified from sensitivity analyses for this receptor						

~~**Notes:**~~

Table 7-21 Notes:
- = not calculated; ND – not detected
NBSA-wide includes exceedances in each of the three zones (N, SE, SW) unless otherwise noted.
*Sediment COPEC concentrations are within background levels and tissue residues are assumed to be related to background conditions as well.

Tissue HQs were estimated for COPECs with sufficient toxicity data to derive TRVs. The dietary model was used to evaluate COPECs for which tissue concentrations may not be useful for evaluation of toxicity, such as some actively-regulated metals and organics subject to metabolism.

The long-term fish community surveys (USEPA 2015) found that indicators of fish community health in Newark Bay provide evidence for a normally functioning fish community. Adult fish abundance and diversity in the Arthur Kill/Newark Bay were similar or greater than other locations in the NY/NJ Harbor Estuary. Juveniles, larvae and eggs, including American eel, were commonly collected in the NBSA and are indicative of ongoing reproduction within fish communities.

The fish pathology evaluation of American eel found only a low percentage of minor pathologies, and does not indicate impairment of fish populations.

Tissue HQs inherently integrate many factors related to fish COPEC exposure, such as site use and dietary composition, typically resulting in a low degree of uncertainty in the tissue EPCs. Toxicity inputs to the tissue HQs, such as the constituent-specific TRVs, are discussed below as part of the WOE evaluation for these receptors. For dietary HQs, additional inputs to the HQs, including SUFs, EPCs, and TRVs, are also discussed as part of the WOE.

SUF: American eel are found within the NBSA during most sampled months and were evaluated for exposure to the NBSA-wide area as well as the individual assessment zones in the dietary models. Although this species spawns in the open ocean, eels have been reported nearly year-round in the NBSA. The selected SUF of 1 is also appropriate for the NBSA-wide area based on the reported home range of about 3 to 65 hectares (Appendix B) for these fish. The SUF of 1 likely overestimates eel exposure in the individual assessment zones.

Diet: In this BERA, eels were assumed to have a mixed diet consisting of invertebrates and fish. These opportunistic carnivores have a diverse diet, which changes from primarily insects to fish and crustaceans as eels grow. The assumed diet for small eels (<50 cm) may overestimate exposure because it does not include a high proportion of insects, and instead includes infaunal invertebrates (i.e., polychaete worm) with high potential for COPEC uptake. The assumed diet for larger eels (≥50 cm) is not expected to over- or underestimate exposure for this receptor in the dietary food web models.

EPCs: In general, EPCs were based on 95% UCL concentrations, which are appropriate exposure estimates for a BERA. A maximum EPC is a less robust exposure estimate for this BERA than EPCs

based on a 95% UCL. In cases where a COPEC is infrequently detected, use of the maximum EPC potentially overestimates exposure. EPC factors considered in the WOE include:

- Methylmercury in worm prey tissue: Methylmercury was detected in only one of eight polychaete worm samples measured as part of the bioaccumulation investigation. The methylmercury EPC for this prey tissue is based on the maximum detected concentration.

Conclusions: For most preliminary COCs identified for this receptor, whole-body tissue HQs were considered to have the least uncertainty for drawing risk conclusions. The dietary models have additional uncertainties related to the use of exposure assumptions (SUFs, diet, BAFs) and the egg HQs were considered less certain due to the use of a CF to estimate egg concentrations. For copper, dietary HQs were considered to be the least uncertain, due to the high degree of uncertainty in the tissue TRVs. Based on the various factors considered in the available dietary and tissue LOEs, the resulting risk conclusions for American eel (<50 cm and ≥50 cm) preliminary COCs are as follows:

- Dioxins/furans and TEQs: Unacceptable risk is considered ~~unlikely~~possible. These conclusions are based on ~~1)~~ tissue LOAEL HQs ~~< 1~~approaching 10 based on ~~NBSA TRVs~~the LPR FFS TRVs. Other available LOEs indicate that unacceptable risk is unlikely for forage fish populations. These include 1) tissue LOAEL HQs < 1 based on NBSA TRVs; 2) fish community surveys do not indicate unhealthy fish populations; and 3) fish pathology does not indicate unhealthy fish populations. Tissue HQs were the only HQ LOE evaluated for this COPEC.

- Total PCB TEQ: Unacceptable risk is considered unlikely. These conclusions are based on 1) tissue LOAEL < 1 for most scenarios; 2) low ~~confidence in~~magnitude of the tissue LOAEL HQs ≥ 1 (HQ = 1 for North Zone only using LPR FFS ~~HQs;~~TRVs); 3) fish community surveys do not indicate unhealthy fish populations; and 4) fish pathology does not indicate unhealthy fish populations. Tissue HQs were the only HQ LOE evaluated for this COPEC.

- Total PCBs: Unacceptable risk is considered unlikely. These conclusions are based on 1) tissue LOAEL HQs < 1 based on NBSA TRVs; 2) low magnitude of the tissue LOAEL HQs ≥1 based on LPR FFS HQs; 3) fish community surveys do not indicate unhealthy fish populations; and 4) fish pathology does not indicate unhealthy fish populations. Tissue HQs were the only HQ LOE evaluated for this COPEC.

- ~~confidence in the LPR FFS HQs; 3) fish community surveys do not indicate unhealthy fish populations; and 4) fish pathology does not indicate unhealthy fish populations. Tissue HQs were the only HQ LOE evaluated for this COPEC.~~

- Total DDx: Unacceptable risk is considered unlikely. These conclusions are based on 1) ~~all~~ tissue LOAEL HQs < 1 ~~based on NBSA TRVs; 2) tissue LOAEL HQs< 1 based on LPR FFS HQs in the~~for most ~~representative exposure area (NBSA-wide area); 3~~scenarios; 2) low magnitude of the LOAEL HQs ≥1 (HQ = 1 in Southwest Zone using LPR FFS TRVs~~; 4) dietary LOAEL HQs < 1; 5~~ only); 3) fish community surveys do not indicate unhealthy fish populations; and ~~6~~4) fish pathology does not indicate unhealthy fish populations.

- Cadmium: Unacceptable risk is considered unlikely. These conclusions are based on 1) cadmium was never detected in whole-body tissue; 2) low magnitude of the dietary LOAEL HQs ≥ 1; 3) conservative exposure assumptions in the dietary models (assumed diet and dietary EPCs likely

overestimate exposure; SUF in assessment zones overestimates exposure); 4) fish community surveys do not indicate unhealthy fish populations; and 5) fish pathology does not indicate unhealthy fish populations.

- Copper: Unacceptable risk is considered unlikely. Although there is low confidence in both the tissue and dietary TRVs, the selected TRVs likely overestimate toxicity of copper. These conclusions are based 1) all tissue LOAEL HQs < 1 using NBSA TRVs; 2) dietary and tissue LOAEL HQs < 1 in the NBSA-wide exposure area using both sets of TRVs; 3) low magnitude of the tissue LOAEL HQs ≥ 1 using LPR FFS TRVs; 4) conservative exposure assumption in the dietary models assumptions (assumed diet likely overestimate exposures for eels < 50 cm; SUF in assessment zones overestimates exposure); 5) fish community surveys do not indicate unhealthy fish populations; and 6) fish pathology does not indicate unhealthy fish populations.
- Mercury and methylmercury: Unacceptable risk is considered unlikely. These conclusions are based on 1) ~~all~~ tissue ~~and dietary~~ LOAEL HQs < 1 ~~using NBSA TRVs~~for most scenarios; 2) dietary and tissue LOAEL HQs < 1 in the NBSA-wide exposure area using both sets of TRVs; 3) low magnitude of the tissue LOAEL HQs ≥ 1 in the North and Southwest Zones ~~using LPR FFS TRVs~~; 3) use of conservative exposure assumptions in the dietary models (assumed diet likely overestimates exposures for eels < 50 cm; SUF in assessment zones overestimates exposure); 4) fish community surveys do not indicate unhealthy fish populations; and 5) fish pathology does not indicate unhealthy fish populations.
- Selenium: Unacceptable risk is considered unlikely. These conclusions are based on 1) tissue LOAEL HQs < 1 for most scenarios; 2) low magnitude of the tissue LOAEL HQs ≥ 1(in the Southeast Zone only); 3) dietary LOAEL HQs < 1; 4) use of conservative exposure assumptions in the dietary models (assumed diet likely overestimates exposures for eels < 50 cm; SUF in assessment zones overestimates exposure); 5) fish community surveys do not indicate unhealthy fish populations; and 6) fish pathology does not indicate unhealthy fish populations. Additionally, selenium concentrations in NBSA sediment are at or below background concentrations.

~~8.7.3~~7.7.3 Summer and Winter Flounder

For summer and winter flounder, whole-body tissue HQs and dietary HQs (evaluated combined summer/winter flounder) were the available HQ LOEs. Fish community surveys (USACE 2015a, 2015b) and fish pathology were considered as indicators of overall fish population health. Background sediment concentrations (Section 4.4.2) were also considered as a LOE. LOAEL HQs ≥ 1 were predicted for the COPECs in Table 7-22. Sensitivity analyses included evaluation of NDs for tissue and dietary HQs, modified prey size assumptions for the dietary HQs, and evaluation of an SUF of 1 for the dietary HQs.

Table 7-22. Summary of LOAEL HQs ≥ 1 for Flounder

Preliminary COCs	Tissue LOAEL HQs ≥ 1		Dietary LOAEL HQs ≥ 1	
	NBSA TRVs	LPR FFS TRVs	NBSA TRVs	LPR FFS TRVs
Primary Exposure and Toxicity Assumptions (Tables 7-8 and 7-13)				
Dioxins/Furans (2,3,7,8-TCDD, Total	None	NBSA-wide, N, SE, SW zones	-	-

Dioxin TEQ Fish, Total Dioxin/PCB TEQ Fish)				
Total PCBs	None	N zone	-	-
Cadmium	-ND	-ND	NBSA-wide, SE, SW zones	-
Copper	None	N zone	None	-
Mercury	None	None	NBSA-wide, N, SW zones	
Sensitivity Analyses (Table 7-18)				
Cadmium	-	-	NBSA-wide, N, SE, SW zone (SUF=1)	None
Mercury	-	-	NBSA-wide, N, SE, SW zone (SUF=1)	None
Methylmercury	-	-	N, SE zones (SUF=1)	None

Notes:
- = not calculated; ND – not detected
NBSA-wide includes exceedances in each of the three zones (north, southeast, southwest) unless otherwise noted.

Tissue HQs were estimated for COPECs with sufficient toxicity data to derive TRVs. The dietary model was used to evaluate COPECs for which tissue concentrations may not be useful for evaluation of toxicity, such as some actively-regulated metals and organics subject to metabolism.

The long-term fish community surveys (USEPA 2015) found that indicators of fish community health in Newark Bay provide evidence for a normally functioning estuarine fish community. Specifically, for winter flounder, adult fish abundance in the Arthur Kill/Newark Bay, were similar or greater than other locations in the NY/NJ Harbor Estuary. Although winter flounder juveniles, larvae and eggs were not as commonly collected in the NBSA relative to other Upper and Lower Bay locations, these differences may be in part explained by physical differences such as bottom substrate, water temperature, salinities, and effects of freshwater inputs that can influence the presence of certain species. Summer flounder adults and larvae/eggs were much less common than winter flounder in NY Harbor but were captured in similar numbers across sampling locations. Overall, the data suggest healthy adult populations and ongoing reproduction of winter flounder in the NBSA.

The fish pathology evaluation included examination of 25 flounder (20 summer and five winter flounder). No pathologies/abnormalities were found in any of these fish.

Tissue HQs inherently integrate many factors related to fish COPEC exposure, such as site use and dietary composition, typically resulting in a low degree of uncertainty in the tissue EPCs. Toxicity inputs to the tissue HQs, such as the constituent-specific TRVs, are discussed above as part of the WOE evaluation for these receptors. For dietary HQs, additional inputs to the HQs, including SUFs, EPCs, and TRVs, are also discussed as part of the WOE.

SUF: Summer and winter flounder are found within the NBSA seasonally. Flounder exhibit strong seasonal inshore-offshore movements, and are found in the Bay generally between March and October, and April and October, respectively (see Section 2.4). The selected SUF of 67% is not expected to significantly over- or underestimate exposure for summer and winter flounder based on seasonal use of Newark Bay for the NBSA-wide area. Because foraging range with flounder when present within the NBSA is unknown, it is uncertain whether the selected SUF overestimates exposure for flounder in the individual assessment zones.

Diet: Flounder are omnivorous or opportunistic feeders, and consume a wide variety of prey found in or near bottom sediments. Diet composition is related to size: the larger the individual, the greater the size of the food item to be consumed. The assumed diet of 35% polychaete worms, 35% blue crabs, and 30% fish (≤ 30 cm) used to represent the prey consumed by flounder is not expected to significantly over- or underestimate exposure for summer and winter flounder, though proportions of these items in their diet may change from juvenile to adult.

EPCs: In general, EPCs were based on 95% UCL concentrations, which are appropriate exposure estimates for a BERA. A maximum EPC is a less robust exposure estimate for this BERA than EPCs based on a 95% UCL. In cases where a COPEC is infrequently detected, use of the maximum EPC potentially overestimates exposure. EPC factors considered in the WOE include:

- Tissue EPCs: for all preliminary COCs in the individual assessment zones, tissue EPCs were based on the maximum detected concentration due to small sample size (N=2 to 4).
- Cadmium in prey fish tissue: Cadmium was not detected in forage fish tissue; the ~~reporting~~quantitation limit was selected as the EPC and likely overestimates exposure to dietary cadmium via consumption of fish.

~~**Conclusions**:~~**Conclusions**: For most preliminary COCs identified for this receptor, whole-body tissue HQs were considered to have the least uncertainty for drawing risk conclusions. The dietary models have additional uncertainties related to the use of exposure assumptions (SUFs, diet, BAFs) and the egg HQs were considered less certain due to the use of a CF to estimate egg concentrations. For copper, dietary HQs were considered to be the least uncertain, due to the high degree of uncertainty in the tissue TRVs. Based on the various factors considered in the available dietary and tissue LOEs, the resulting risk conclusions for summer/winter flounder preliminary COCs are as follows:

- Dioxin/furans and TEQs: Unacceptable risk is considered ~~unlikely in the NBSA or individual assessment zones.~~possible. These conclusions are based on tissue LOAEL HQs approaching 10 based on the LPR FFS TRVs, particularly in the North Zone. Other available LOEs indicate that unacceptable risk is unlikely for forage fish populations. These include 1) all tissue LOAEL HQs < 1 based on NBSA TRVs; 2) ~~low confidence in the LPR FFS HQs; 3) fish community surveys do not indicate unhealthy fish populations; and 4) fish pathology does not indicate unhealthy fish populations. Tissue HQs were the only HQ LOE evaluated for this COPEC.~~
- fish community surveys do not indicate unhealthy fish populations; and 3) fish pathology does not indicate unhealthy fish populations. Tissue HQs were the only HQ LOE evaluated for this COPEC.
- Total PCBs: Unacceptable risk is considered unlikely ~~in the north assessment zone and NBSA-wide.~~. These conclusions are based on 1) tissue LOAEL HQs < 1 based on NBSA TRVs; 2) low ~~confidence in the FFS HQs; 3) low~~ magnitude of the LOAEL HQ ≥ 1 using LPR FFS TRVs~~; 4~~ (HQ = 1 in the North Zone only); 3) EPCs in the individual assessment zones based on maximum detected concentrations; ~~5~~4) fish community surveys do not indicate unhealthy fish populations; and ~~6~~5) fish pathology does not indicate unhealthy fish populations. Tissue HQs were the only HQ LOE evaluated for this COPEC.
- Cadmium: Unacceptable risk is considered unlikely in the southeast and southwest assessment zones and NBSA-wide. These conclusions are based on 1) low magnitude of the dietary LOAEL HQs

≥ 1; 2) cadmium was never detected in flounder or prey fish tissue; 3) fish community surveys do not indicate unhealthy fish populations; and 4) the fish pathology does not indicate unhealthy fish populations.

- Copper: Unacceptable risk is considered unlikely in the north assessment zone and NBSA-wide. Although there is low confidence in both the tissue and dietary TRVs, the selected TRVs likely overestimate toxicity of copper. These conclusions are based on 1) all tissue LOAEL HQs < 1 using NBSA TRVs; 2) all dietary and tissue LOAEL HQs < 1 in the NBSA-wide exposure area using both sets of TRVs; 3) low magnitude of the tissue LOAEL HQs ≥ 1 using LPR FFS TRVs; 4) maximum tissue EPCs in the assessment zones; 5) fish community surveys do not indicate unhealthy fish populations; and 6) fish pathology does not indicate unhealthy fish populations.

~~8.7.4~~7.7.4 White Perch

For white perch, whole-body tissue HQs, liver HQs, and dietary HQs were the available HQ LOEs. LOAEL HQs ≥ 1 were predicted as shown in Table 7-23. Fish community surveys (USACE 2015a, 2015b) and fish pathology were considered as indicators of overall fish population health. Background sediment concentrations (Section 4.4.2) were also considered as a LOE. LOAEL HQs ≥ 1 were predicted for the COPECs listed in Table 7-23. Sensitivity analyses included evaluation of NDs for tissue and dietary HQs, evaluation of an SUF of 1 for the dietary HQs, and the evaluation of an alternative TRV for fish tissue dioxin/furan TEQ.

Table 7-23. Summary of LOAEL HQs ≥ 1 for White Perch

Preliminary COCs	Whole-body LOAEL Tissue HQs ≥ 1		Liver LOAEL HQs ≥ 1	Dietary LOAEL HQs ≥ 1	
	NBSA TRVs	LPR FFS TRVs	NBSA Liver TRVs	NBSA TRVs	LPR FFS TRVs
Primary Exposure and Toxicity Assumptions (Tables 7-8 and 7-13)					
Dioxins/Furans (2,3,7,8-TCDD, Total Dioxin .Furan TEQ)	NBSA-wide, N, SE, SW zones*** ~~None~~	NBSA-wide, N, SE, SW zones	None	-	-
Total PCBs	None	NBSA-wide, N, SE, SW zones	None	-	-
Total PCB TEQ	None	N zone	None	-	-
Total DDx	NBSA-wide, SW zone~~None~~	NBSA-wide, N, SE, SW zones	-~~None~~	-	-
Arsenic	None	-	NBSA-wide, SW zone	None	-
Cadmium	-ND	-ND	-	NBSA-wide, N, SE, SW zones	-
Chromium	N zone	-	NBSA-wide, SW zone	None	-
Copper**	NBSA-wide, N, SE, SW zones	NBSA-wide, N, SE, SW zones	NBSA-wide, N, SE, SW zones-	None	-
~~Mercury~~	~~None~~	~~None~~	~~None~~	~~None~~	~~-~~
Selenium*	SW zone~~None~~	-	NBSA-wide, N, SE, SW zones	None	-
Silver*	NBSA-wide, N, SE, SW zones	-	NBSA-wide, N, SE, SW zones	None	-
Zinc	None	-	NBSA-wide, N, SE, SW zones	None	-
Sensitivity Analyses (Table 7-18)					
Dioxins/Furans (2,3,7,8-TCDD, Total Dioxin/Furan TEQ)	NBSA-wide, N, SE, SW zones (alternative TRV)	-	-	-	-
Mercury	-	-	-	NBSA-wide, SE, SW zones (SUF=1)	None

Notes:
- ~~Not~~not calculated; ND – not detected
~~NBSA-wide includes exceedances in each of the three zones (N, SE, SW) unless otherwise noted.~~
*Sediment COPEC concentrations are within background levels and tissue residues are assumed to be related to background conditions as well.
**Tissue residues in mummichog were within the range of Jamaica Bay background residues; white perch residues also assumed to be background-related.
*** Based on alternate TRV for TEQ.

Tissue HQs were estimated for COPECs with sufficient toxicity data to derive TRVs. The dietary model was used to evaluate COPECs for which tissue concentrations may not be useful for evaluation of toxicity, such as some actively-regulated metals and organics subject to metabolism.

The long-term fish community surveys (USEPA 2015) found that indicators of fish community health in Newark Bay provide evidence for a normally functioning fish community. Adult white perch abundance in the Arthur Kill/Newark Bay, was greater than other locations in the NY/NJ Harbor Estuary during both winter and spring surveys. White perch eggs and larvae were rarely captured during ichthyoplankton surveys primarily because this species spawns upstream from Newark Bay.

Although white perch were not specifically examined in the fish pathology evaluation, only a low percentage of minor pathologies were found across all species. These results do not indicate impairment of fish populations. The conclusions likely also apply to white perch.

Tissue HQs inherently integrate many factors related to fish COPEC exposure, such as site use and dietary composition, typically resulting in a low degree of uncertainty in the tissue EPCs. Toxicity inputs to the tissue HQs, such as the constituent-specific TRVs, are discussed below as part of the WOE evaluation for these receptors. For dietary HQs, additional inputs to the HQs, including SUFs, EPCs, and TRVs, are also discussed as part of the WOE.

SUF: Multi-year fish sampling data indicate that white perch are seasonally abundant in Newark Bay in late fall and winter (October through May; Appendix B) only. White perch are not present in late spring, summer, and early fall due to spawning migration into streams feeding Newark Bay. The selected SUF of 67% is not expected to significantly over- or underestimate exposure for white perch based on seasonal use of Newark Bay for the NBSA-wide area. Because foraging range of white perch when present within the NBSA is unknown, it is uncertain whether the selected SUF overestimates exposure for fish in the individual assessment zones.

Diet: White perch are invertivorous benthic-feeding fish; common dietary components include amphipods, shrimp, and copepods. Fish are also a dietary component, with a greater proportion of fish consumed in late summer and fall. The assumed diet of 70% polychaete worms (a surrogate for amphipods and other invertebrates), 15% blue crabs, and 15% fish (≤ 30 cm) used to represent the prey consumed by white perch potentially overestimates exposure because polychaete worms likely have higher potential for COPEC uptake than small, epibenthic crustaceans.

EPCs: In general, EPCs were based on 95% UCL concentrations which are appropriate exposure estimates for a BERA. A maximum EPC is a less robust exposure estimate for this BERA than EPCs based on a 95% UCL. In cases where a COPEC is infrequently detected, use of the maximum EPC potentially overestimates exposure. EPC factors considered in the WOE include:

- Liver EPCs: for all preliminary COCs in the individual assessment zones, liver tissue EPCs were based on the maximum detected concentration due to small sample size. Liver EPCs for chromium in the NBSA-wide area were also based on the maximum detected concentration due to low FOD.
- Polychaete worm prey tissue EPCs: The worm tissue EPCs evaluated in the assessment zones are based on NBSA-wide EPCs derived from the worm bioaccumulation investigation data.

~~**Conclusions**:~~**Conclusions**: For most preliminary COCs identified for this receptor, whole-body tissue HQs were considered to have the least uncertainty for drawing risk conclusions. The dietary models have additional uncertainties related to the use of exposure assumptions (SUFs, diet, BAFs) and the egg HQs were considered less certain due to the use of a CF to estimate egg concentrations. For copper, dietary HQs were considered to be the least uncertain, due to the high degree of uncertainty in the tissue TRVs.

For white perch, the liver HQs were not used to draw risk conclusions in the WOE because whole-body HQs were also available for the same COPECs, and are a more relevant indicator of overall exposure. The liver assessment was used to provide perspectives on the whole-body risk findings. Based on the various factors considered in the available dietary and whole-body tissue LOEs, the resulting risk conclusions for white perch preliminary COCs are as follows:

- Dioxin/furans and TEQs: Unacceptable risk is considered ~~unlikely in the NBSA or individual assessment zones.~~possible. These conclusions are based on 1) ~~all~~ tissue LOAEL HQs ~~< ~~>1 for all scenarios; and 2) tissue HQs > 10 based on ~~NBSA TRVs; 2) low confidence in the~~ LPR FFS TRVs~~; 3)~~ . Other available LOEs (fish community surveys and fish pathology) do not indicate ~~unhealthy fish populations; and 4) fish pathology does not indicate unhealthy fish populations.~~the potential for unacceptable risk.
- Total PCB TEQ: Unacceptable risk is considered unlikely. These conclusions are based on 1) tissue LOAEL HQs < 1 for most scenarios; 2) low magnitude of the tissue LOAEL HQs ≥ 1 (HQ =1 in the North zone using LPR FFS TRVs only); 3) fish community surveys do not indicate unhealthy fish populations; and 4) fish pathology does not indicate unhealthy fish populations.
- Total PCBs: Unacceptable risk is considered unlikely in the NBSA or individual assessment zones. These conclusions are based on 1) all tissue LOAEL HQs < 1 based on NBSA TRVs; 2) low ~~confidence in the FFS HQs; 3) low~~ magnitude of the tissue LOAEL HQ ≥ 1 using LPR FFS TRVs; 4) fish community surveys do not indicate unhealthy fish populations; and 5) fish pathology does not indicate unhealthy fish populations.
- Total DDx: Unacceptable risk is considered ~~unlikely in the NBSA or individual assessment zones.~~possible. These conclusions are based on ~~1) all~~ tissue LOAEL HQs ~~<~~≥ 1 ~~based on~~for most scenarios using NBSA ~~TRVs; 2) low confidence in the~~and LPR FFS ~~HQs; 3) low magnitude of the tissue LOAEL HQ ≥ 1 using LPR FFS TRVs; 4)~~ TRVs. The HQs do not exceed 2 in any case, and other available LOEs (fish community surveys and fish pathology) do not indicate ~~unhealthy fish populations; and 5) fish pathology does not indicate unhealthy fish populations~~the potential for unacceptable risk.
- Cadmium: Unacceptable risk is considered unlikely in all assessment zones and NBSA-wide. These conclusions are based on 1) ~~low magnitude of the dietary LOAEL HQs ≥ 1; 2)~~ cadmium was never detected in perch or prey fish tissue; ~~3~~2) low magnitude of the dietary LOAEL HQs ≥ 1; 3) dietary assumptions likely overestimate exposure (e.g., use of polychaete worm to represent unidentified dietary material); 4) fish community surveys do not indicate unhealthy fish populations; and ~~4~~5) the fish pathology does not indicate unhealthy fish populations.
- Chromium: Unacceptable risk is considered unlikely~~ in the north assessment zone and NBSA-wide~~. There is low confidence in both the tissue and dietary TRVs. The dietary LOAEL TRV is based on an unbounded NOAEL and the tissue LOAEL TRV is based on tissue data unrelated to the observed toxicity; both TRVs likely overpredict risk. LPR FFS TRVs were not available for chromium. The risk conclusion is based on 1) all dietary LOAEL HQs < 1; 2) tissue LOAEL HQs < 1 ~~for~~except in the ~~NBSA-wide area;~~North Zone (HQ = 1); 3) low magnitude of the tissue LOAEL HQs ≥ 1; 4) fish community surveys do not indicate unhealthy fish populations; and 5) fish pathology does not indicate unhealthy fish populations.

- Copper: Unacceptable risk is considered uncertain in the NBSA and individual assessment zones. There is low confidence in both the tissue and dietary TRVs, and the selected TRVs likely overestimate toxicity of copper. Because the tissue TRVs are well below nutritional thresholds for copper, the dietary NBSA HQs are considered to have lower uncertainty for predicting risk than the tissue HQs. However, there is also only moderate confidence in the dietary HQs to predict risk to fish due the compounded uncertainty in various exposure and toxicity assumptions included in the dietary models. Risk is considered uncertain for copper are based on: 1) all dietary LOAEL HQs < 1; 2) all tissue LOAEL HQs ≥ 1 using NBSA and LPR FFS TRVs and tissue LOAEL HQs approaching or exceeding 10 in many cases, 3) maximum tissue EPCs are used to assess risk in the assessment zones; 4) fish community surveys do not indicate unhealthy forage fish populations; and 5) fish pathology does not indicate unhealthy fish populations. In addition, the copper concentrations in NBSA forage fish were at or below background concentrations for mummichog collected from Jamaica Bay (by NOAA in 1999), and, therefore, it is reasonable to assume that white perch concentrations are also likely within background. However, given that background data do not exist for white perch, the risks, while unlikely, are considered uncertain.
- Mercury: Unacceptable risk is considered unlikely in the NBSA or individual assessment zones. These conclusions are based on 1) tissue LOAEL HQs < 1 all scenarios; 2) low magnitude of the dietary LOAEL HQs ≥ 1 (maximum HQ =2 in the sensitivity analyses); 3) use of conservative exposure assumptions in the dietary models (assumed diet likely overestimates exposures); 4) fish community surveys do not indicate unhealthy fish populations; and 5) fish pathology does not indicate unhealthy fish populations.
- Silver: Unacceptable risk is considered unlikely in the NBSA or individual assessment zones. These conclusions are based on 1) dietary LOAEL HQs < 1 in the NBSA-wide area; 2) low magnitude of the tissue LOAEL HQs ≥ 1; 3) fish community surveys do not indicate unhealthy forage fish populations; and 4) fish pathology does not indicate unhealthy fish populations. Although the whole-body tissue and dietary silver TRVs are uncertain, the risk conclusions are supported by additional LOE, including sediment concentrations in the NBSA are at or below background concentrations and liver HQs based on a NOAEL TRV are low in magnitude (liver LOAEL TRV unavailable for silver).
- Other metals (arsenic~~, mercury~~, selenium, and zinc): Unacceptable risk is not predicted in the NBSA or individual assessment zones for most metals because whole-body tissue and dietary LOAEL HQs <1 were predicted. As noted above, although liver LOAEL HQs ≥ 1 were estimated for some metals, these HQs were based on NOAELs, and are superseded by whole-body tissue HQs, which do not confirm the liver TRV findings.

~~8.7.5~~7.7.5 Summary of WOE Risk Findings for Fish

Overall, unacceptable risk is ~~not~~ predicted for fish potentially exposed to a limited number of COPECs in the NBSA. ~~The LOE~~Based on the WOE, potential for ~~the various~~unacceptable risk was identified for dioxins/furans and TEQ (2,3,7,8-TCDD, total dioxin/furan TEQ fish, dioxin/furan/PCB TEQ fish) and all fish receptors~~ and COPECs, and overall risk conclusions based on the WOE are summarized~~, total PCBs for forage fish in ~~Table 7-24~~the Southwest Zone only, and ~~discussed below~~total DDx for white perch. Potential risk from copper is considered uncertain for white perch, and this COPEC was recommended as a final COC due to the substantial uncertainty associated with the risk estimates for copper. The LOE for

the various fish receptors and COPECs, and overall risk conclusions based on the WOE are summarized in Table 7-24 and discussed below. Fish community monitoring data from the ABS (USACE 2015a, 2015b) found that fish abundance and diversity in the Arthur Kill/Newark Bay were similar or greater than other locations in the NY/NJ Harbor Estuary, and are comparable. Additionally, for all fish evaluated in this BERA, the pathology investigation indicates healthy fish populations within the NBSA. The fish community surveys and pathology investigations support the conclusion of no unacceptable risk presented below ~~based on available HQ LOEs.~~for most COPECs and receptors. Conclusions for the fish evaluations are based on the WOE evaluation and are summarized below:

- For localized forage fish (mummichog/killifish):
  - ~~Tissue, dietary, and egg LOAEL HQs < 1 were estimated for most COPECs and exposure areas using NBSA TRVs, except for cadmium, copper, and silver. Using LPR FFS TRVs, whole-body tissue LOAEL HQs ≥ 1 were estimated for TEQ, total PCBs, total DDx, and copper.~~
  - Preliminary COCs included dioxins/furans and TEQ (2,3,7,8-TCDD, total dioxin/furan TEQ fish, dioxin/furan/PCB TEQ fish), total PCB TEQ, total PCBs, total DDx, cadmium, copper, mercury, methylmercury, and silver (based on sensitivity analyses only) based on tissue, dietary, and/or egg LOAEL HQs ≥ 1 estimated for at least one scenario evaluated in this BERA.
  - Unacceptable risk is considered possible for dioxins/furans and TEQ, and for total PCBs in the Southwest Zone only.
    - For dioxins/furans and TEQ, the conclusion of unacceptable risk is based primarily on tissue LOAEL HQs greater than 10 based on the LPR FFS TRVs and tissue LOAEL HQs greater than 1 based on the alternative NBSA TRV. Other available LOEs (tissue HQs based on NBSA TRVs, egg HQs, and fish pathology) indicate that unacceptable risk is unlikely for ~~organics (TEQ, total PCBs~~forage fish populations.
    - For total PCBs, unacceptable risk is considered unlikely except possibly in the Southwest Zone. These conclusions are based on LOAEL tissue HQs below or near 1 for most scenarios, except in the Southwest Zone where the HQ = 6 using LPR FFS TRVs. Egg LOAEL HQs are below 1 even using a conservative CF =1; and fish pathology does not indicate unhealthy forage fish populations.
  - Unacceptable risk is considered unlikely for the remaining preliminary COCs (total PCB TEQ, total DDx~~)~~, and some metals ~~(~~[cadmium, copper, mercury, methymercury, and silver~~).~~]):
    - For organics (~~TEQ,~~ total ~~PCBs,~~PCB TEQ and total DDx), these conclusions are based primarily on ~~moderate confidence in the NBSA~~ whole-body tissue LOAEL HQs < 1 for most scenarios and low ~~confidence in~~magnitude of the tissue LOAEL HQs ≥ 1 using LPR FFS ~~tissue~~ TRVs~~, which likely overpredict risk associated with exposure to these COPECs~~. Egg tissue LOAEL HQs < 1 were also predicted for ~~TEQ and~~ total ~~PCBs~~PCB TEQ using conservative adult-to-egg transfer assumptions, including with the LPR FFS TRVs, which were derived from an egg-based assessment.
    - For cadmium, these conclusions are based on higher confidence in the tissue LOE in which cadmium was not detected in whole-body fish tissue and liver NBSA LOAEL HQs were < 1. The dietary LOAEL HQs ≥ 1 are low in magnitude.

- For ~~silver~~copper, these conclusions are based on ~~the tissue and~~ conflicting conclusions from two uncertain HQs methods (dietary ~~NBSA LOAEL HQs < 1 in the NBSA-wide area, low magnitude of the tissue LOAEL HQ ≥ 1, and sediment silver concentrations below background.~~

- ~~o~~ ~~Unacceptable risk is considered unlikely for copper. Both NBSA and LPR FFS tissue LOAEL HQs ≥ 1 are predicted for the NBSA-wide area and all zones (Southeast and Southwest Zones for LPR FFS LOAEL HQs only); whereas dietary NBSA LOAEL HQs <1 were estimated using conservative dietary assumptions. There~~and tissue HQs). There is substantial uncertainty related to both the tissue and dietary TRVs; although, both sets of HQs likely overestimate risk. Although tissue LOAEL HQs ≥ 1 were predicted, dietary HQs were < 1, incorporate conservative assumptions (100% worm diet) and were consistent with other LOE, including that fish pathology does not indicate unhealthy forage fish populations. In addition, as reported in the LPRSA BERA (Windward 2019), the background concentration of copper in mummichog collected from Jamaica Bay (by NOAA in 1999) was 4,200 µg/kg. The NBSA forage fish concentrations are at or below this concentration.

- Mercury and methylmercury: Unacceptable risk is considered unlikely. These conclusions are based on whole-body tissue and dietary LOAEL HQs < 1 for most scenarios, low magnitude of the dietary LOAEL HQs, equal to 1 in the Southwest Zone for mercury only, and egg LOAEL HQs < 1 even using a conservative CF =1.

- For silver, these conclusions are based on sediment silver concentrations at or below background. Additionally, tissue and dietary LOAEL HQs < 1 were predicted for most scenarios except a tissue LOAEL HQ equal to 1 in the North Zone. Fish pathology does not indicate unhealthy fish populations.

- For predatory fish (American eel, flounder, and white perch):
  - Fish community monitoring data for these three species (USACE 2015a, 2015b) found no impairment relative to other locations in the NY/NJ Harbor Estuary.
  - ~~Tissue and/~~Fish pathology data do not indicate unhealthy fish populations in the NBSA.
  - Preliminary COCs, identified based on tissue or dietary LOAEL HQs ~~<~~≥ 1 ~~were estimated~~ for ~~most COPECs and exposure areas using NBSA TRVs, except for chromium for white perch, cadmium for~~ any scenarios evaluated in this BERA, include:
    - For American eel ~~and summer/winter flounder, copper for white perch, and silver for white perch. Using LPR FFS TRVs, whole-body tissue LOAEL HQs ≥ 1 were estimated for all predatory fish for some organics (~~: dioxins/furans and TEQ (2,3,7,8-TCDD, total dioxin/furan TEQ fish, dioxin/furan/PCB TEQ fish), total PCB TEQ, total PCBs, total DDx~~) and a few metals (copper for white perch, flounder, and American eel and~~ , cadmium, copper, mercury~~/~~, methylmercury~~ for American eel~~, and selenium.
    - ~~o~~ For summer and winter flounder: dioxins/furans and TEQ (2,3,7,8-TCDD, total dioxin/furan TEQ fish, dioxin/furan/PCB TEQ fish), total PCBs, cadmium, copper, mercury, and methylmercury (based on sensitivity analyses only).

- ~~Unacceptable risk is considered unlikely for organics (~~For white perch: dioxins/furans and TEQ (2,3,7,8-TCDD, total dioxin/furan TEQ fish, dioxin/furan/PCB TEQ fish), total PCB TEQ, total PCBs, total DDx~~) and some metals (~~, cadmium, chromium, copper, mercury (based on sensitivity analyses only), selenium, and silver. Additionally, arsenic and zinc were identified as preliminary COCs based only on liver LOAEL HQs ≥ 1; whole body tissue and dietary LOAEL HQs <1 were predicted for arsenic and zinc in all scenarios.

o Unacceptable risk is considered possible for dioxin/furans and TEQ (all predatory receptors) and for total DDx (white perch only).

- For dioxins/furans and TEQ, the conclusion of unacceptable risk is based primarily on whole-body tissue LOAEL HQs near or greater than 10 for all scenarios based on the LPR FFS TRVs. For white perch, whole body tissue LOAEL HQs ≥ 1 were also predicted in some cases using NBSA TRVs; however, tissue LOAEL HQs < 1 were predicted for eel and flounder. Other available LOEs (tissue HQs based on NBSA TRVs, white perch liver HQs, fish community data, and fish pathology) indicate that unacceptable risk is unlikely for fish populations.

- For total DDx, unacceptable risk was considered unlikely for American eel, but possible for white perch. Total DDx was not identified as a preliminary COC for summer/winter flounder. For eel, tissue LOAEL HQs < 1 were predicted in most cases except in the Southwest Zone using LPR FFS TRVs (HQ =1). For white perch, tissue HQs ≥ 1 were predicted in most cases; however, the magnitude of the HQs is low (HQ ≤ 2). Other available LOEs (fish community data and fish pathology) indicate that unacceptable risk is unlikely for fish populations.

o Unacceptable risk is considered unlikely for the remaining preliminary COCs (total PCB TEQ, total PCBs and some metals [arsenic, cadmium, chromium, mercury/methylmercury, ~~and~~selenium, silver~~):~~, and zinc]):

- For organics (total PCB TEQ~~,~~ and total PCBs~~, total DDx~~), these conclusions are based primarily on ~~moderate confidence in the NBSA~~ whole-body tissue LOAEL HQs < 1 in most cases and low ~~confidence in the~~magnitude of the tissue LOAEL HQs ≥ 1 (calculated using LPR FFS ~~tissue~~ TRVs~~, which likely overpredict risk associated with exposure to~~ only).

- For arsenic, these ~~COPECs~~conclusions are based on whole body tissue and dietary LOAEL HQs < 1 in all scenarios. Although liver LOAEL HQs ≥ 1 were estimated, these HQs are superseded by whole-body tissue HQs, which do not confirm the liver TRV findings.

- For cadmium, these conclusions are based on higher confidence in the tissue LOE in which cadmium was rarely detected in whole-body fish tissue~~ and liver NBSA LOAEL HQs were < 1.~~. The dietary LOAEL HQs ≥ 1 are low in magnitude~~.~~. Cadmium was not detected in most whole body fish tissue (only in a single flounder sample) and therefore the tissue EPCs used to calculated both tissue HQs and prey tissue EPCs in the dietary HQs likely overestimate exposure.

- For chromium, there is low confidence in both the tissue and dietary TRVs, which both likely overpredict risk. The risk conclusion is based on 1) all dietary LOAEL HQs < 1; 2) tissue

LOAEL HQs < 1 for ~~the NBSA-wide area and all predatory fish~~ most scenarios (except white perch in the north assessment zone~~;~~); 3) low magnitude of the LOAEL tissue HQs ≥ 1.

- For mercury/methylmercury, these conclusions are based on ~~high confidence in the~~ tissue ~~NBSA and LPR FFS~~ LOAEL HQs~~, which are less than~~ < 1 ~~for all~~in most scenarios ~~using NBSA tissue and~~, dietary ~~TRVs. Using LPR FFS tissue TRVs,~~ LOAEL HQs < 1 ~~were estimated for the NBSA-wide area, and LOAEL HQs ≥ 1 for the North and Southwest Zones are~~ in most scenarios, and low ~~in~~ magnitude of the LOAEL HQs ≥ 1.
- For selenium, these conclusions are based on tissue LOAELs < 1 in most scenarios, dietary LOAEL HQs < 1 in all scenarios, and low magnitude of the tissue LOAEL HQs ≥ 1 (HQ = 1~~).~~ for eel in the Southeast Zone and white perch in the Southest Zone). In addition, selenium concentrations in NBSA sediment are likely related to background.
- For silver, these conclusions are based on dietary LOAEL HQs <1 and the low magnitude of the tissue LOAEL HQs, which are greater than 1 only for white perch (whole-body). There is low confidence in the available silver TRVs. Liver HQs for silver based on an unbounded NOAEL are low in magnitude and sediment silver concentrations are less than background. The HQs greater than 1 likely reflect uncertainty in the TRVs and silver exposures in the NBSA are considered to be related to background conditions.
- For zinc, these conclusions are based on whole body tissue and dietary LOAEL HQs <1 in all scenarios. Although liver LOAEL HQs ≥ 1 were estimated, these HQs are based on NOAEL TRVs and are superseded by whole-body tissue HQs, which do not confirm the liver TRV findings.

o For copper, unacceptable risk is considered uncertain for white perch and unlikely for American eel and summer/winter flounder. For white perch, ~~both NBSA and LPR FFS~~ tissue LOAEL HQs ≥ 1 are predicted ~~for the NBSA-wide area~~in all scenarios and ~~all zones; whereas dietary NBSA~~the tissue LOAEL HQs approach or exceed 10 in many cases; however, dietary LOAEL HQs <1 were estimated using conservative dietary assumptions. There is substantial uncertainty related to both the tissue and dietary TRVs; although, both sets of HQs likely overestimate risk. For American eel and summer/winter flounder, NBSA tissue and dietary LOAEL HQs < 1 were estimated and LPR FFS tissue LOAEL HQs ≥ 1 were low in magnitude and were estimated only in the north assessment zone (flounder) or southwest assessment zone (eel).

Based on the results of this BERA, potential exposures to most COPECs in the NBSA are not expected to pose unacceptable risk to fish. ~~The only LOAEL TRV exceedances found other than copper were using the LPR FFS TRVs as discussed above. These results are more uncertain and do not align with the NBSA TRV-based results or other LOEs.~~ As summarized in Table 7-24, the potential for unacceptable risk (identified based on LOAEL HQs ≥1 and WOE supporting the conclusion of unacceptable risk) was ~~not identified.~~identified for dioxin/furans and related TEQ compounds (2,3,7,8-TCDD, total dioxin/furan TEQ fish, dioxin/furan/PCB TEQ fish) for all fish receptors, total PCBs for forage fish in the Southwest Zone, and possibly for total DDx for white perch in the NBSA. Unacceptable risk is uncertain for white perch exposed to copper.

~~9 BIRD ASSESSMENT~~

8 BIRD ASSESSMENT

This section includes the risk assessment for birds in the NBSA. Birds are part of the upper trophic levels of the food web in the Bay. There are three feeding guilds of birds evaluated in this BERA: benthivores (consume benthic invertebrates), piscivores (consume fish), and omnivores (consume plant and animal matter). The representative bird receptors for each of these feeding guilds are listed in Table 3-2 and include spotted sandpiper, great blue heron, double-crested cormorant, and lesser scaup.

The following are the Assessment ~~Endpoints~~Endpoint and Risk ~~Questions~~Question for birds that are addressed in this section.

Assessment Endpoint: Survival, growth, and/or reproduction of birds.

Risk Question 1: *Does the daily dose of COPECs received by birds (including piscivorous, benthivorous/sediment-probing, omnivorous, invertivorous birds) from consumption of the tissues of prey species and from other media in the NBSA exceed the TRVs for survival, growth, and/or reproduction of birds?*

Risk Question 1 is addressed by the bird dietary risk assessment and bird egg tissue assessment presented in Section 8.1 and 8.2, respectively. The life history profiles for the bird receptors identified for the tissue and dietary risk assessments are provided in Appendix B. An uncertainties assessment is provided for the overall bird assessment in Section 8.3, and the WOE summary of findings for birds is provided in Section 8.4.

The species selected for assessment in this BERA were identified from historical surveys (Section 2.9) as predominant birds that forage in the NBSA, and some of which may breed in wetlands within the NY/NJ Harbor Estuary, as well as elsewhere within their home range. These birds include species from different feeding guilds, and with different levels of potential exposure to COPECs in sediments and the food web of the system. For these reasons, the assessment provided in this section is fully representative of the potential risks to populations of birds that utilize the NBSA at variable times in their life histories.

~~9.1~~8.1 Dietary Assessment

A dietary assessment was conducted for each of the four selected bird species (spotted sandpiper, great blue heron, double-crested cormorant, and lesser scaup). This section describes how exposure and effects concentrations were derived and presents the HQs for the dietary assessment. Section 8.3 presents the uncertainties associated with the dietary assessment for birds. The COPECs evaluated for the bird dietary assessment are listed in Table 3-5 and include: 2,3,7,8-TCDD, total dioxin/furan TEQ, total PCBs (209 congeners), total PCB TEQ, total dioxin/furan/PCB TEQ, pesticides (total chlordane, total DDx, dieldrin, and hexachlorobenzene), total HMW/LMW PAHs and total PAHs, and inorganics (arsenic, cadmium, chromium, copper, lead, mercury/methylmercury, nickel, selenium, silver, and zinc).

~~9.1.1~~8.1.1 Exposure

This section presents the methods for calculating exposure doses, including descriptions of the selection of parameters for the dose equation, composition of prey in diet, exposure areas, and EPCs in prey. The exposure factors used for the bird dietary assessment are presented in Table 8-1.

~~9.1.1.1~~8.1.1.1 Dietary Dose Model

Dietary doses for birds were estimated based on ingestion of biota (i.e., prey) and the incidental ingestion of sediment. Surface water ingestions was not included in bird dietary assessment because birds were not expected to drink the saltwater of the NBSA. Dietary doses were estimated as milligrams of each COPEC ingested per kilogram of body weight per day using the following equation:

$$ADD_{total} = [(EPC_s * SIR/BW * DF_s) + \Sigma(EPC_k * FIR_k/BW * DF_k)] * SUF$$ ***Equation 8-1***

where,

ADD_{total} = average daily dose (mg/kg bw/day)

EPC_s = exposure point concentration in sediment (mg/kg, dw)

SIR = ingestion rate for sediment (kg/day, dw)

BW = body weight of organism (kg)

DF_s = dietary fraction of sediment ingested (range 0 to 1)

EPC_k = EPC for the kth food type (mg/kg day, ww)

FIR_k = ingestion rate for the kth food type (kg/day, ww)

DF_k = dietary fraction of intake of the kth food type (range 0 to 1)

SUF = site use factor (unitless); proportion of time the selected species spends foraging in the NBSA

~~9.1.1.2~~8.1.1.2 Body Weights, Ingestion Rates, Prey Composition, Site Use Factors, Exposure Areas

Body weights and ingestion rates were obtained from the literature for each representative species being modelled. Body weights and ingestion rates for each receptor are presented in Table 8-1. The dietary fraction of each component in each species' diet is based on information obtained from the literature (Appendix B) and are presented in Table 8-1. Prey ingested by bird species were limited to whole-body samples only from prey types for which empirical tissue chemistry data from the NBSA were available (polychaete worm, blue cram, softshell clam, and fish). Great blue heron, double-crested cormorant, and lesser scaup were assumed to consume fish ≤ 15 cm as part of their diet. A sensitivity analysis was also performed to evaluate consumption of larger fish up to 30 cm in length, as described in Section 8.3.

The diet of the spotted sandpiper consists primarily of terrestrial and aquatic invertebrates. Examples of prey consumed includes flies, grasshoppers, crickets, beetles, caterpillars, worms, spiders, fish, crustaceans, and mollusks (Reed et al. 2013). Spotted sandpipers foraging in the NBSA were assumed to have a diet of 100% polychaete worm to represent their invertebrate prey items.

Great blue heron diet is primarily fish, but they may also consume amphibians, reptiles, crustaceans, insects, birds, and mammals (USEPA 1993). Based on an average beak length of 13.5 cm (Vennesland and Butler 2011), Krebs (1974) determined that more than 92% of great blue heron fish prey are small or medium sized (up to about 6.8 cm in length), and the remaining fish prey are greater than or equal to the length of the beak (13.5 cm). Fish up to about 20 cm in length were dominant in the diet of herons foraging in southwestern Lake Erie and 95 percent of fish consumed by in a Wisconsin population were less than 25 cm in length (USEPA 1993). Based on the information from Krebs (1974), fish ≤ 15 cm were included as 80% of the diet for the NBSA. Because great blue heron are opportunistic foragers and will feed on prey other than fish, the remaining 20% of the diet was assumed to be 10% crab and 10% clam.

Double-crested cormorants eat almost entirely fish, usually slow-moving or schooling species (Dorr et al. 2014). Prey size ranges from 3 to 40 cm, but most commonly less than 15 cm. Therefore, for the NBSA, a diet of 100% fish (≤ 15 cm) was assumed.

Specific components of the lesser scaup diet in winter vary geographically and often reflect local abundance (Anteau et al. 2014). In the winter in the Gulf of Mexico, scaup feed nearly exclusively on surf clams (*Mulinia lateralis*) and other mollusks (Harmon 1962); when wintering in coastal bays and marshes, they feed mainly on midges (46% of total dry weight) and bulrush seeds (36%) (Afton et al. 1991); along the Columbia River in the Pacific Northwest, most of the diet is vegetation (Thompson et al. 1988); and in the Midwest, Jones and Drobney (1986) have reported that scaup consume oligochaete worms in the Detroit River. During the winter, there are no significant differences in diet between juveniles and adults or between males and females (Anteau et al. 2014). Based on a diet of primarily aquatic invertebrates, the diet evaluated in the NBSA included 25% blue crab, 25% softshell clam, 25% polychaete worms, and 25% fish (≤ 15 cm).

While fish, blue crab, and softshell clam data were field collected, the worm tissue data were based on the laboratory bioaccumulation study in which worms were exposed to homogenized sediment collected from the 0 to 15-cm depth horizon.

The SUFs for the representative bird receptors are provided in Table 8-1. Spotted sandpiper is migratory and is expected in Newark Bay only during the breeding season (Reed et al. 2013) and was given an SUF of 0.5. Both great blue heron and double-crested cormorant are expected in Newark Bay year-round (Vennesland and Butler 2011; Dorr et al. 2014) and were given SUFs of 1.0. Lesser scaup breeds in the boreal forests of northern North America and is expected in Newark Bay only during the winter or non-breeding season (Anteau et al. 2014) and was therefore given an SUF of 0.5.

The exposure areas for the receptor bird species are provided in Table 3-3. Spotted sandpiper and great blue heron risk from prey consumption was evaluated NBSA-wide and by assessment zone (north, southeast, southwest) and risk from incidental sediment ingestion was evaluated using sediment samples from NBSA-wide mudflats and mudflats within the individual assessment zones. Risk (from prey and incidental sediment ingestion) to double-crested cormorant and lesser scaup were evaluated NBSA-wide and by assessment zone. Spotted sandpiper and lesser scaup included polychaete worm as a portion of their diet; however, polychaete worm sample size was not large enough to be evaluated by assessment zone. Therefore, NBSA-wide UCL EPCs were used to represent polychaete worm inputs to the food web models. See Tables 4-10 and Figure 4-4 for sediment samples assigned to each assessment zone and mudflats, and Table 4-11 for tissue samples assigned to each assessment zone.

~~9.1.1.3~~8.1.1.3 Exposure Point Concentrations

EPCs were calculated for each of the two modeled media types (prey and sediment) to calculate dietary doses using Equation 8-1. For prey concentrations, EPCs were calculated separately for each of the prey types for each species (worms, blue crab, softshell clam, fish ≤ 15 cm) as the UCL for each prey group. EPCs for prey were calculated NBSA-wide and by assessment zone (except polychaete worm, where EPCs were calculated NBSA-wide only due to limited sample size). For sediment, the EPCs were equal to the UCLs using data from the relevant exposure areas for each bird species. ~~UCL concentrations were calculated using USEPA's ProUCL® statistical package (Version 5.1) (USEPA 2015b).~~UCL concentrations were calculated using USEPA's ProUCL® statistical package (Version 5.1) (USEPA 2015b).[10] For each dataset with fewer than six detected samples or fewer than six total samples, a UCL was not calculated; instead, the maximum detected concentration was used as the EPC. If no samples were detected, then the maximum non-detect value was used as the EPC. The sediment and biota tissue data used in the bird dietary assessment are summarized in Section 4.4. The EPCs calculated for the bird dietary assessment are presented in Appendix C (Table C-1 for sediment, C-2 for fish ≤ 15 cm, C-3 for fish ≤ 30 cm, C-4 for polychaetes, C-5 for softshell clam, and C-6 for blue crab).

~~9.1.1.4~~8.1.1.4 Estimated Doses

Dietary doses were calculated based on Equation 8-1 using the prey and sediment ingestion rates, species body weights, and the prey composition from Table 8-1 and the EPCs from Appendix C. These doses are presented in Appendix E.

~~9.1.2~~8.1.2 Effects

Dietary bird TRVs for the COPECs identified in Table 3-5 are provided in Table 8-2, and discussed in detail in Appendix D. The selected TRVs were primarily derived from the LPRSA BERA TRV database (Windward 2019~~).~~) and are consistent with the Final LPRSA BERA TRVs for COPECs common to both areas (NBSA and LPRSA). For the LPRSA BERA, Windward (2019) conducted a literature search for relevant toxicological studies, and derived acceptable NOAELs and LOAELs. The lowest acceptable LOAEL and highest NOAEL below the LOAEL from the same study were selected. Several NBSA COPECs were not evaluated in the LPRSA BERA bird dietary risk assessment including arsenic, silver, dieldrin, total chlordane, and hexachlorobenzene. TRVs for arsenic, dieldrin, total chlordane, and hexachlorobenzene were included in the LPRSA BERA and SLERA TRV databases (Windward 2019), and were evaluated in the TRV selection process for this BERA. A silver TRV based on USEPA (1997) was selected. The selected NBSA TRVs for these additional COPECs are further discussed in Appendix D.

In addition, bird dietary TRVs from the LPR FFS (USEPA 2014) were also evaluated and are included in Table 8-2. The LPR FFS bird dietary TRVs are available for 2,3,7,8-TCDD, total PCBs, total DDx, dieldrin, LMW PAH, HMW PAH, copper, lead, and mercury/methylmercury. They were not available for total chlordane, hexachlorobenzene, arsenic, cadmium, chromium, nickel, selenium, silver, zinc, or total PAH.

[10] The UCL recommended by USEPA's ProUCL® software (typically the 95% UCL, but in some cases the 97.5% or even the 99% UCL) was used. The selected UCL statistic is presented in Appendix C-1.

~~9.1.3~~8.1.3 Risk Characterization

This section presents the comparison of bird dietary doses to NBSA BERA and LPR FFS TRVs to calculate HQs for spotted sandpiper, great blue heron, double-crested cormorant, and lesser scaup. NBSA-wide NOAEL and LOAEL HQs for the bird dietary assessment based on NBSA BERA TRVs are presented in Table 8-3a. Assessment zone (i.e., north, southeast, and southwest) NOAEL and LOAEL HQs based on NBSA BERA TRVs are presented in Tables 8-3b through 8-3d. LPR FFS HQs are also presented in Tables 8-3a through 8-3d; Appendix E provides calculated HQs for the each COPEC and bird species. A summary of COPECs that exceed their TRVs (i.e., HQs that equal or exceed 1) is provided in Table 8-4.

LOAEL HQs ≥ 1 were estimated for one or more avian receptors for ~~lead, mercury, methylmercury,~~ total PCB TEQ bird, total dioxin/furan/PCB ~~Dioxin~~ TEQ bird, ~~and~~ total DDx~~.~~, lead, and mercury/methylmercury. For ~~copper, zinc, total HMW PAH,~~ 2,3,7,8-TCDD, total dioxin/furan TEQ bird, total ~~PCB Dioxin TEQ bird, total~~ PCBs, total HMW PAH, and ~~total chlordane,~~copper NOAEL HQs ≥ 1 and LOAEL HQs < 1 were estimated. The remaining COPECs (i.e., most inorganics/metals [arsenic, cadmium, chromium, nickel, selenium, ~~and~~ silver, and zinc], LMW PAHs, total PAHs, dieldrin, total chlordane, and hexachlorobenzene) resulted in NOAEL HQs < 1 for all bird receptors evaluated.

Table 8-4. Summary of TRV Exceedances for Bird Dietary Assessment

~~Constituents~~	~~NBSA HQs ≥ 1~~															
	~~Spotted Sandpiper~~								~~Great Blue Heron~~							
	~~NBSA-wide~~		~~North~~		~~SE~~		~~SW~~		~~NBSA-wide~~		~~North~~		~~SE~~		~~SW~~	
	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~
~~Dioxins/Furans~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~Total Dioxin/Furan TEQ Bird~~	-	○	-	○	-	○	-	○	-	○	-	○	-	○	-	○
~~PCBs~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~Total PCBs~~	-	-	-	-	-	-	-	-	○	-	○	-	-	-	○	-
~~Total PCB TEQ Bird~~	-	○	-	○	-	○	-	○	○	○	○	○	○	○	○	○
~~Total Dioxin/Furan/ PCB TEQ Bird~~	-	○	-	○	-	○	○	○	○	○	○	○	○	○	○	○
~~Pesticides~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~Total DDx (2,4 & 4,4)~~	-	-	-	-	-	-	-	-	○	●	○	●	○	●	○	●
~~Total Chlordane~~	○	--	○	--	○	--	○	--	○	--	○	--	○	--	○	--
~~PAHs~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~Total HMW PAH~~	--	○	--	-	--	○	--	○	--	-	--	-	--	-	--	-
~~Inorganics~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~Copper~~	-	-	-	-	-	-	○	-	-	-	-	-	-	-	-	-
~~Lead~~	-	●	-	○	-	○	○	●	-	○	-	○	-	○	-	○
~~Mercury~~	-	○	-	○	-	○	-	●	-	○	-	○	-	○	-	-
~~Methylmercury~~	-	-	-	-	-	-	-	-	-	○	-	○	-	○	-	○

~~Zinc~~	-	--	-	--	-	--	⊖	--	-	--	-	--	-	--	-	--

~~Constituents~~	~~NBSA HQs ≥ 1~~															
	~~Double-Crested Cormorant~~								~~Lesser Scaup~~							
	~~NBSA-wide~~		~~North~~		~~SE~~		~~SW~~		~~NBSA-wide~~		~~North~~		~~SE~~		~~SW~~	
	~~NBSA~~	~~LPR-FFS~~	~~NBSA~~	~~LPR-FFS~~	~~NBSA~~	~~LPR-FFS~~	~~NBSA~~	~~LPR-FFS~~	~~NBSA~~	~~LPR-FFS~~	~~NBSA~~	~~LPR-FFS~~	~~NBSA~~	~~LPR-FFS~~	~~NBSA~~	~~LPR-FFS~~
~~**Dioxins/Furans**~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~Total Dioxin/Furan TEQ Bird~~	-	⊖	-	⊖	-	⊖	-	⊖	-	-	-	⊖	-	-	-	-
~~**PCBs**~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~Total PCBs~~	⊖	-	⊖	-	⊖	-	⊖	⊖	-	-	-	-	-	-	-	-
~~Total PCB TEQ Bird~~	⊖	⊖	⊖	⊖	⊖	⊖	⊖	●	-	⊖	-	⊖	-	⊖	-	⊖
~~Total Dioxin/Furan/ PCB TEQ Bird~~	⊖	●	⊖	●	⊖	●	⊖	●	-	⊖	-	⊖	-	⊖	-	⊖
~~**Pesticides**~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~Total DDx (2,4 & 4,4)~~	⊖	●	⊖	●	⊖	●	⊖	●	-	-	-	⊖	-	-	-	⊖
~~Total Chlordane~~	⊖	--	⊖	--	⊖	--	⊖	--	⊖	--	⊖	--	⊖	--	⊖	--
~~**PAHs**~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~Total HMW PAH~~	--	-	--	-	--	-	--	-	--	-	--	-	--	-	--	-
~~**Inorganics**~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~Copper~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~Lead~~	-	-	-	⊖	-	-	-	-	-	⊖	-	⊖	-	⊖	-	●
~~Mercury~~	-	⊖	-	⊖	-	⊖	-	-	-	-	-	-	-	-	-	-
~~Methylmercury~~	-	⊖	-	⊖	-	●	-	⊖	-	-	-	-	-	-	-	-
~~Zinc~~	-	--	-	--	-	--	-	--	-	--	-	--	-	--	-	--

Constituents	NBSA HQs ≥ 1															
	Spotted Sandpiper								Great Blue Heron							
	NBSA-wide		North		SE		SW		NBSA-wide		North		SE		SW	
	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS
Dioxins/Furans																
2,3,7,8-TCDD		○		○		○		○		○		○		○		
Total Dioxin/Furan TEQ Bird		○		○		○		○		○		○		○		○
Polychlorinated Biphenyls																
Total PCBs									○		○				○	
Total PCB TEQ Bird		○		○		○		○	○	○	○	○	○	○	○	○
Total Dioxin/Furan/PCB TEQ Bird		○		○		○	○	○	○	○	○	○	○	○	○	○
Pesticides																
Total DDx (2,4 & 4,4)									○	●	○	●	○	●	○	●
Dieldrin																
Total Chlordane		--		--		--		--		--		--		--		--
Hexachlorobenzene		--		--		--		--		--		--		--		--
Polycyclic Aromatic Hydrocarbons																
Total HMW PAH	--	○	--		--	○	--	○	--		--		--		--	
Total LMW PAH	--		--		--		--		--		--		--		--	
Total PAH		--		--		--		--		--		--		--		--
Inorganics																
Arsenic		--		--		--		--		--		--		--		--
Cadmium		--		--		--		--		--		--		--		--
Chromium		--		--		--		--		--		--		--		--
Copper							○									
Lead		●		○		○	○	●		○		○		○		○
Mercury	○	○	○	○	○	○	○	●	○	○	○	○	○	○	○	
Methylmercury									○	○	○	○	○	○	○	○
Nickel		--		--		--		--		--		--		--		--
Selenium		--		--		--		--		--		--		--		--
Silver		--		--		--		--		--		--		--		--
Zinc		--		--		--		--		--		--		--		--

Notes: ● = indicates LOAEL HQ ≥ 1; ○ = indicates NOAEL HQ ≥ 1; -- = not calculated; COPECs evaluated that did not have any exceedances are not shown.

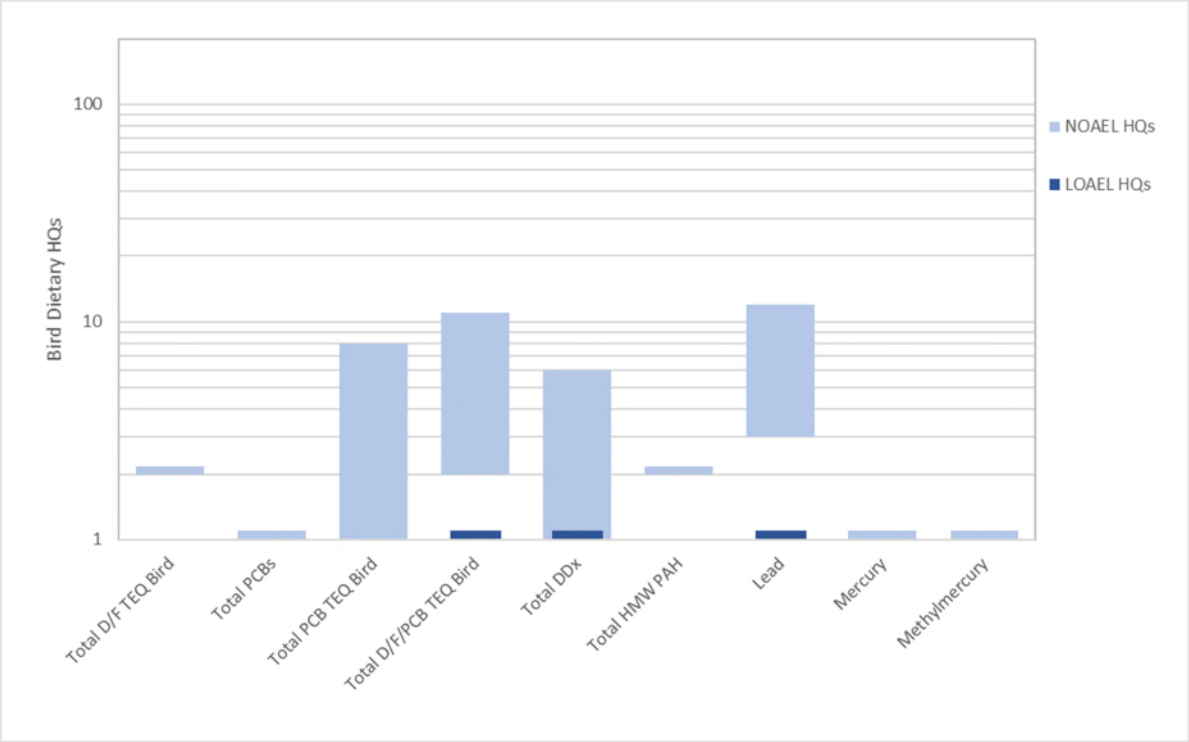
Table 8-4. Summary of TRV Exceedances for Bird Dietary Assessment (cont.)

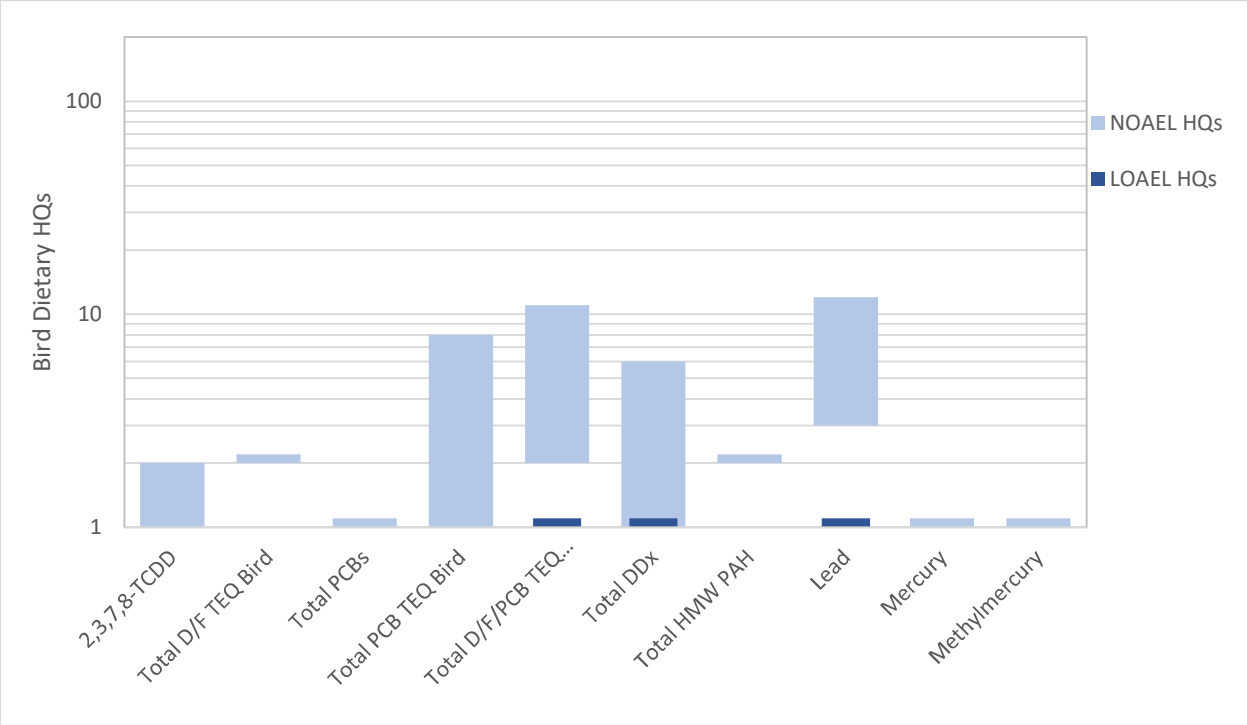
Constituents	NBSA HQs ≥ 1															
	Double-Crested Cormorant								Lesser Scaup							
	NBSA-wide		North		SE		SW		NBSA-wide		North		SE		SW	
	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS
Dioxins/Furans																
2,3,7,8-TCDD		○		○		○		○								
Total Dioxin/Furan TEQ Bird		○		○		○		○				○				
Polychlorinated Biphenyls																
Total PCBs	○		○		○		○	○								
Total PCB TEQ Bird	○	○	○	○	○	○	○	●		○		○		○		○
Total Dioxin/Furan/PCB TEQ Bird	○	●	○	●	○	●	○	●		○		○		○		○
Pesticides																
Total DDx (2,4 & 4,4)	○	●	○	●	○	●	○	●				○				○
Dieldrin																
Total Chlordane		--		--		--		--		--		--		--		--
Hexachlorobenzene		--		--		--		--		--		--		--		--
Polycyclic Aromatic Hydrocarbons																
Total HMW PAH	--		--		--		--		--		--		--		--	
Total LMW PAH	--		--		--		--		--		--		--		--	
Total PAH		--		--		--		--		--		--		--		--
Inorganics																
Arsenic		--		--		--		--		--		--		--		--
Cadmium		--		--		--		--		--		--		--		--
Chromium		--		--		--		--		--		--		--		--
Copper																
Lead				○						○		○		○		●
Mercury	○	○	○	○	○	○	○				○					
Methylmercury	○	○	○	○	○	●	○	○								
Nickel		--		--		--		--		--		--		--		--
Selenium		--		--		--		--		--		--		--		--
Silver		--		--		--		--		--		--		--		--
Zinc		--		--		--		--		--		--		--		--

Notes: ● = indicates LOAEL HQ ≥ 1; ○ = indicates NOAEL HQ ≥ 1; -- = not calculated; COPECs evaluated that did not have any exceedances are not shown.

The range of HQs for the aggregate bird dietary exposure estimates (i.e., all species) with HQs ≥ 1 are depicted in Figure 8-1.

Figure 8-1. NBSA-wide Magnitude of NOAEL and LOAEL HQs ≥ 1 for Bird Dietary Assessment





~~9.2~~8.2 Egg Tissue Assessment

The potential risks to bird eggs from maternal dietary exposure were evaluated for piscivorous birds, represented by the double-crested cormorant, in the NBSA. A NBSA-specific study was conducted by the Manomet Center for Conservation Science in 1999 (Parsons 2003) on contaminants in bird eggs and measures of related reproductive success on Shooters Island, the only known bird breeding site in the Bay. This study is provided as Attachment 6 to this BERA.

Parsons (2003) analyzed samples of double-crested cormorant eggs, blood, and feathers for various chemical parameters (e.g., dioxins/furans, PCBs, pesticides, PAHs, and metals) from three islands throughout the NY/NJ Harbor Estuary, including Shooters Island in the Bay. The objectives of the study were to determine if concentrations of chemicals were present in the birds at levels of concern, and to determine if there was any correlation between the concentrations of the chemicals and reproductive success in the population.

For the BERA bird egg tissue assessment, double-crested cormorant egg tissue data collected by Parsons (2003) were compared to available bird-egg based tissue TRVs from the literature. In addition, the COPECs analyzed in the study for which egg TRVs are not available were assessed in the context of the study findings with respect to the measured reproductive endpoints, most notably hatching and fledgling success of the eggs/fledglings. COPECs for piscivorous bird egg tissue are listed in Table 3-5 and include: 2,3,7,8-TCDD, total dioxin/furan TEQ, total PCBs~~ (209 congeners),~~, total PCB TEQ, total dioxin/furan/PCB TEQ, pesticides (total chlordane, total DDx, dieldrin, and hexachlorobenzene), ~~total PAHs, cadmium,~~ and mercury/methylmercury.

~~9.2.1~~8.2.1 Exposure

COPEC concentrations in bird eggs were taken from the Parsons (2003) study of double-crested cormorants in Newark Bay. In 1999, egg samples were collected from a double-crested cormorant colony located on Shooters Island in Newark Bay. Reproductive endpoints were assessed in a total of 61 nests in that colony and 20 of those nests were chosen for egg sample collection. One intact, newly laid egg was collected from the selected study nests in each colony and were analyzed for dioxins/furans, PCBs, chlorinated pesticides, PAHs, cadmium, and mercury/methylmercury. Eggs were also collected from two other islands, Gardiner's Island, located in Long Island Sound and considered a control, and Swinburne Island, located in lower New York Harbor for comparison.

Double-crested cormorant egg tissue data used in this evaluation were taken from the data tables in Parsons (2003). Total dioxin/furan TEQ and total dioxin/furan/PCB TEQ were calculated using the methods described in Section 4.2.3 using the data provided in Parsons (2003). Summary COPEC concentrations and selected EPCs for the bird egg assessment are presented in Appendix C.

~~9.2.2~~8.2.2 Effects

Bird egg TRVs for dioxins/furans, PCBs, total DDx, and mercury/methylmercury are provided in Table 8-5 and discussed in detail in Appendix D. These TRVs were primarily derived from the LPRSA BERA TRV database (Windward 2019~~).~~) and are consistent with the Final LPRSA BERA TRVs. In addition, bird egg TRVs from the LPR FFS (USEPA 2014) were also included for comparison with the NBSA-selected

TRVs. LPR FFS bird egg TRVs were not available for mercury/methylmercury or total PCBs. Bird egg TRVs were not found for other pesticides such as chlordane, dieldrin, or hexachlorobenzene. Further discussion of the bird egg TRVs is in Appendix D.

In addition to the TRV-based assessment, the field-based study results presented in Parsons (2003) are included to assess the potential risks from the various COPECs measured in the study. This is an important line of evidence for this BERA, as the study evaluated the nesting and reproductive success of the birds investigated and measured their actual exposure (i.e., egg concentrations) to the COPECs in the system.

~~9.2.3~~8.2.3 Risk Characterization

Bird egg HQs were calculated from data in Parsons (2003) and the NBSA TRVs identified in Table 8-5. Bird egg HQs are presented in Table 8-6, and the full HQ calculations are presented in Appendix E. LOAEL bird egg HQs were ≥ 1 using both the NBSA and LPR FFS TRVs for the following COPECs: ~~Total~~total PCBs, ~~Total~~total PCB TEQ, ~~Total~~ total dioxin/furan/PCB ~~Dioxin/Furan~~ TEQ~~. Total~~, and total DDx. Mercury and methylmercury LOAEL ~~HQ was >~~HQs were ≥1 for the LPR FFS TRV only. For 2,3,7,8-TCDD and total dioxin/furan TEQ, NOAEL HQs ≥ 1 and LOAEL HQs < 1 were estimated.

Tissue samples collected from birds on Shooters Island generally contained higher concentrations of COPECs than tissues collected from birds on the other two islands (Swinburne and Gardiner's Islands). However, no correlation was found between double-crested cormorant reproductive success (e.g., egg production, hatching success, and chick survival) and COPEC concentrations among the three islands (Table 8-7; Parsons 2003). Eggs from many of the study nests on both Gardiner's and Shooters Islands failed to hatch as a result of being lost from the nest—suggesting predation was a major cause of nest failure. No differences were detected among islands in the proportion of young that died in or near the nest (typically considered to be inviable nestlings) and the proportion of young missing from the nest (considered to have been predated). The authors themselves noted that there were no associations between the COPECs measured in bird eggs (or blood and feathers) and reproductive success. Given that chemical contamination levels in the NBSA (as measured in sediments by the USEPA's REMAP between 1993 and 2013; Appendix A) have declined in the past two decades, it is likely that COPEC exposures of birds foraging in the Bay ~~to COPECs has~~have also declined. The Parsons (2003) study, as such, likely represents a conservative estimate of potential present-day risks.

Table 8-7. Reproductive Endpoints for Double-Crested Cormorant Examined in Parsons (2003)

Reproductive Endpoint	Location	Number of Nests	Significance (Chi-square test)	Conclusion
Hatching Success	Gardiners Island (Control)	14	P=0.001	No impacts to Shooters Island hatching success relative to control
	Shooters Island	38		
	Swinburne island	3		
Hatching Failure	Gardiners Island (Control)	16		
	Shooters Island	3		
	Swinburne island	0		
Fledging Success	Gardiners Island (Control)	5	P=0.0007	

	Shooters Island	30	No impacts to Shooters Island fledge success relative to control
	Swinburne island	24	
Fledging Failure	Gardiners Island (Control)	9	
	Shooters Island	8	
	Swinburne island	3	

~~9.3~~8.3 Uncertainties Assessment

This section contains the uncertainties assessment for the bird risk assessment. Both quantitative and qualitative characterizations are provided for the dietary and egg assessments presented in Sections 8.1 and 8.2, respectively. The objective of the uncertainty assessment is to provide perspective on the likely accuracy of the overall risk conclusions that are presented in Section 8.4.

Sources of uncertainty that influenced bird risk characterization included uncertainties in the analytical results, data evaluation, CSM, exposure assessment, effects assessment, and interpretation of the risk estimates to make conclusions regarding potential for unacceptable population-level risk. Many of these sources of uncertainty are generic in nature and inherent in the risk assessment process.

Inherent practices in risk assessment result in exposure estimates that are biased high in order to be reasonably conservative include:

- The use of an upper exposure value (i.e., 95% UCL) to calculate dietary exposure concentrations
- The use of a maximum values in the place of a UCL when limited data are available
- The assumption that a bird species feeds exclusively or is getting all of its seasonal and lifetime exposure to COPECs, from the NBSA (i.e., SUF = 1)
- The assumption that COPECs are 100% bioaccessible via dietary uptake (dermal absorption via the gut).

As a result, risk estimates are expected to be overestimated rather than underestimated in this BERA.

For risk estimates based on dietary and egg tissue HQs, TRVs are a key source of uncertainty in the risk estimates and are discussed below or in Appendix D, as relevant. Additional sources of uncertainty were identified and evaluated either quantitatively or qualitatively. Table 8-8 presents a summary of the quantitative uncertainty evaluations conducted for birds for the dietary and egg assessments and their associated effect on risk estimates. Risk calculations and HQs based on the quantitative uncertainty evaluations are presented in Appendix F and discussed in the following sections. Qualitative evaluations were also conducted and are presented below in their respective sections.

Table 8-8. Summary of Quantitative Bird Uncertainty Evaluations

Species	General Uncertainty Evaluated	Specific Evaluation Conducted	Difference in HQ Values	Effect on Risk Conclusions[a,d]
Bird Dietary Assessment				

Species	General Uncertainty Evaluated	Specific Evaluation Conducted	Difference in HQ Values	Effect on Risk Conclusions[a,d]
Spotted sandpiper	SUF of 0.5	HQs recalculated using SUF of 1	HQs increase (100%)	Total Dioxin/Furan/PCB TEQ Bird LOAEL HQ based on LPR FFS TRVs changed from < 1 to ≥ 1 for SW zone only; mercury LOAEL HQ based on LPR FFS TRVs changed from < 1 to ≥ 1 for NBSA-wide, north, SE zones
Spotted sandpiper; Lesser scaup	NBSA-wide Polychaete worm EPCs used	Representative COPEC re-evaluated using sediment-to-worm regression and zone sediment EPCs	HQs decrease (≤ 0.1)	No change
All	Treatment of NDs as zero for EPC calculation	EPCs recalculated using half or full DL	HQs increase (< 0.2)	Increase in HQs to ≥ 1 for Total Dioxin/Furan/PCB TEQ using ND=1 in NBSA-wide, north, SE zones for sandpiper. Increase in HQs to ≥ 1 for Total HMW PAHs using ND=1/2 and ND=1 in North zone for sandpiper.~~No change~~
Great blue heron	Fish prey size of <15 cm assumed	EPCs recalculated using all prey fish	HQs increase (<1)	N zone: Increase HQs to ≥ 1 for mercury, methylmercury, Total PCB TEQ, and Total Dioxin/Furan/PCB TEQ using LPR FFS TRVs
~~Great blue heron;~~ Double-crested cormorant	Fish prey size of <15 cm assumed	EPCs recalculated using all prey fish	HQs increase (<1)	NBSA-wide: Increase ~~in~~ HQs to ≥ 1 for ~~mercury[b]~~mercury, methylmercury using LPR FFS TRVs N zone: Increase HQs to ≥ 1 for ~~mercury, methylmercury,~~ Total PCB TEQ, ~~and Total Dioxin/Furan/PCB TEQ[c]~~mercury, methylmercury using LPR FFS TRVs SE zone: Increase HQs to ≥ 1 for ~~mercury[b]~~mercury using LPR FFS TRVs SW zone: Increase HQs to ≥ 1 for ~~copper[b], mercury[b], methylmercury[b], Decrease HQs to <1 for~~ Total PCB TEQ ~~Bird[b], and Total Dioxin/Furan/PCB TEQ[b]~~

Species	General Uncertainty Evaluated	Specific Evaluation Conducted	Difference in HQ Values	Effect on Risk Conclusions[a,d]
				~~using LPR FFS TRVs~~ , , copper, mercury, methylmercury
Great blue heron diet	Effect of including fish > 30 cm in dietary risk assessment	Following alternative diets evaluated: A: 100% fish > 30 cm B: 10% clam; 10% crab; 75% fish ≤ 15 cm; 5% fish > 30 cm C: 10% clam; 10% clam; 70% fish ≤ 15 cm; 10% fish > 30 cm	No change	Diet A: NBSA-wide and all zones increase HQs to ≥ 1 for mercury, methylmercury using LPR FFS TRVs. No change for Diets B or C
Lesser scaup	Effect of including fish in dietary risk assessment	Following alternative diets evaluated: A: 25% crab; 25% clam; 50% worm B: 25% crab; 50% clam; 25% worm C: 75% clam; 25% worm	HQs increase (< 0.2)	No change for Diets A or B Diet C; NBSA-wide lead LOAEL HQs based on LPR FFS TRVs increased to HQ ≥ 1 ~~diet was assumed to be 75% clam and 25% worm~~
Bird Egg Assessment				
Double-crested Cormorant	TEQ TRV Based on 5th Percentile SSD	HQs re-calculated using species-specific TRV	HQs decrease 20-40x	LOAEL HQs all < 1

Notes:
a. Based on LOAEL HQs ≥ 1.
b. Double-crested cormorant only.
c. Great blue heron only
d. All results of the sensitivity analyses are provided in Appendix F.

Additionally, relatively large exposure areas were evaluated in this BERA, including the NBSA-wide area and individual assessment zones. While these areas are appropriate for risk assessment, especially for population-level assessment of bird communities, evaluation of smaller features, such as individual wetlands or mudflats, may be required in the FS. The uncertainty associated with applying the results of the NBSA-wide and assessment zone conclusions to localized areas within the NBSA is considered to be low for populations of mobile receptors such as most birds. For species with higher site fidelity (e.g., spotted sandpiper and great blue heron), this uncertainty was reduced by evaluating mudflat areas instead of the larger assessment zones in the mummichog dietary assessment, which assumed incidental ingestion of mudflat sediment and ingestion of the dietary fraction of worm tissue estimated using bioaccumulation testing data collected from mudflat locations.

~~9.3.1~~8.3.1 Bird Dietary Risk Characterization Uncertainties

Dietary exposure modeling has been a well-established practice for wildlife (avian and mammalian) risk assessments (USEPA 1993; Sample et al. 1997) and oral doses are a common method of evaluating contaminant exposure. However, there are specific uncertainties related to the dietary risk estimates for

birds in the NBSA BERA, including factors related to both exposure and effects assumptions as discussed below. Uncertainties that were evaluated quantitatively are summarized above in Table 8-8.

~~9.3.1.1~~8.3.1.1 Avian Toxicology/TRVs

A key uncertainty associated with the avian dietary risk characterization is the selection of TRVs used in the risk calculations. As noted in Section 3.5 and at the request of USEPA, two sets of TRVs were evaluated to maintain consistency and allow comparison between regional BERAs.

In general, dietary TRVs ~~are~~were selected to be protective of either the 5th percentile of a SSD or the lowest acceptable LOAEL based on single studies reviewed. Thus, they tend to err on the side of over predicting risk. As a result, this section focuses on the TRV uncertainty for those COPECs where LOAEL HQs ≥ 1, which was the case only for lead, mercury, methylmercury, total DDx, total PCB TEQ bird, and total PCB dioxin TEQ bird using LPR FFS dietary TRVs. For all COPECs and receptors, dietary LOAEL HQs were less than 1 when NBSA TRVs were used. Each of the NBSA TRVs was the same as was used in the LPRSA BERA (Windward 2019). General uncertainties associated with TRVs for all COPECs are discussed in Appendix D.

The NBSA TRVs for methylmercury and total DDx are considered to be more robust and less uncertain than the LPR FFS TRVs because the NBSA TRVs were based on SSD and those SSDs include species known or potentially resident in the NBSA (mallard, kestrel, great egret, double-crested cormorant). Six species were used in the development of the methylmercury SSD and 10 species were used in the development of the total DDx SSD, whereas the LPR FFS TRVs are based on data for a single species. For methylmercury, the LPR FFS TRVs include an uncertainty factor to account for interspecies sensitivity and are based on a form of mercury unlikely to be present in the NBSA. For total DDx, the LPR FFS TRVs are based on data from a field study with pelicans that does not provide a clear dose-response relationship. For mercury, the NBSA and LPR FFS TRVs evaluated in this BERA are based on the methylmercury TRVs. There is uncertainty in this approach, which is assumed to overestimate toxicity of mercury to birds.

For lead, four studies were found sufficient in the LPRSA for determining a lead dietary TRV ~~in~~for birds, but this was an insufficient number to develop a TRV. The lowest LOAEL (28 mg/kg bw-day) was selected as the LOAEL TRV and represented a 10% decrease in body weight compared to the control where the test species was Japanese quail (Morgan et al. 1975). Because is it not known whether a reduction in body weight is sufficient to alter reproduction or survival, this TRV likely overestimates risk. The TRV used in the LPR FFS was also based on Japanese quail as the test species, but the endpoint evaluated was egg production. Domesticated species such as quail are bred to have very high egg-laying rates, and it is not evident that the same concentration would reflect adverse effects on reproduction in wild birds. Both sets of TRVs are based on Japanese quail exposed to lead acetate, a soluble form of lead that potentially overestimates bioavailability and toxicity of lead in the NBSA.

There are very limited avian toxicity data available for dioxin/furans. A study by Nosek et al. (1992) was selected as the basis for both the LPRSA /NBSA and LPR FFS TRVs. Nosek et al. (1992) measured adverse effects (mortality, growth, and reproduction) in ring-necked pheasants exposed to 2,3,7,8-TCDD via intra-peritoneal injection. The LPRSA/NBSA BERA selected the LOAEL (nanograms per kilogram per day [ng/kg-day]) as the LOAEL TRV, whereas the LPR FFS applied an uncertainty factor of 5 to the LOAEL to account for interspecies sensitivity resulting in a LOAEL TRV of 28 ng/kg-day. The domestic

chicken, which has the Type I AhR receptor (high sensitivity), is the most sensitive species known to TEQ-mediated toxicity. Ring-necked pheasants have moderate sensitivity (Type II AhR receptor) to dioxin-like compounds. Most wildlife species have not been shown to be as sensitive to dioxins/furans as chickens. Specifically, more than 85 wild species have been genetically sequenced to date. Gray catbird is the only wild species tested by Farmahin et al. (2012) and subsequently by Eng et al. (2014) that is identified as high sensitivity and has been observed in the NBSA. Other wild birds known to be present in the NBSA are known (or assumed based on phylogenetic relationships) to have moderate to low sensitivity to the dioxin-like compounds. Thus, application of an uncertainty factor to account for interspecies sensitivity ~~is not warranted.~~likely overestimates risk to most resident bird species in the NBSA. In cases where wild birds have low sensitivity, the selected NBSA LOAEL TRV is expected to overestimate toxicity as well. Both the LPRSA and LPR FFS used an uncertainty factor of 10 to estimate NOAEL TRVs from the LOAEL TRVs.

The lack of available TRVs for some pesticides such as chlordane, dieldrin, and hexachlorobenzene is an uncertainty in the egg assessment, as a HQ analysis cannot be performed. However, pesticide analyses were included in the Parsons (2003) study in the NBSA, and the potential integrated effects of these compounds is assessed as part of that study. As such, an adult bird HQ assessment using literature TRVs would not likely change the outcome of this assessment.

~~9.3.1.2~~8.3.1.2 Exposure Assumptions

In this section, uncertainties specific to the NBSA dietary exposure assessment were identified and were evaluated quantitatively or qualitatively to assess potential impact on the risk estimates presented above.

Uncertainties related to the selected SUFs, polychaete worm EPCs, treatment of non-detects in calculated mixture totals (e.g., total PCBs), and prey fish size were quantitatively evaluated. These uncertainties are discussed below and summarized in Table 8-8. Other uncertainties, discussed below, were qualitatively assessed.

~~9.3.1.2.1~~8.3.1.2.1 Site Use Factors

A SUF of 0.5 was applied to exposure estimates for the spotted sandpiper and lesser scaup to account for seasonal breeding migration for these species. The SUFs were selected based on data from the literature and bird species observed during surveys of the NBSA. The SUF does not account for the species foraging range when present in the NBSA, and therefore potentially overestimates exposure. Use of the SUF is reasonable and appropriate based on the known migration patterns of these species and limited habitat found within the NBSA.

An alternate SUF of 1 was evaluated for spotted sandpiper, which results in additional COPECs with LOAEL HQs ≥ 1. When spotted sandpiper dietary food web models were analyzed using a SUF = 1 rather than 0.5, the Southwest Zone ~~total dioxin/furan~~Total Dioxin/Furan/PCB TEQ bird LOAEL HQ based on LPR FFS TRVs changed from < 1 to ≥ 1. The mercury LOAEL HQs based on LPR FFS TRVs changed from < 1 to ≥ 1 for NBSA-wide, north, and SE zones. ~~Summary tables of HQ~~ The results of this sensitivity analysis are summarized in Table 8-8. Detailed HQ tables from this sensitivity analysis on the effect of the SUF on dietary risk are presented in Appendix F.

~~9.3.1.2.2~~8.3.1.2.2 *Polychaete Worm EPCs*

The polychaete worm tissue EPCs used in the food web models were based on the laboratory bioaccumulation study in which worms were exposed to homogenized sediment collected from the 0 to 15-cm depth horizon. Uncertainties related the representativeness of these bioaccumulation tests for estimating dietary exposures are discussed in Section 7.6.3.3. Spotted sandpiper and lesser scaup included polychaete worm as a portion of their diets, however, polychaete worm tissue sample size was not large enough to be evaluated by assessment zone. Therefore, risk from ingestion of polychaete worms were assessed using the NBSA-wide EPCs in the assessment zone risk calculations for these species.

A regression analysis was performed to assess the relationship between sediment concentrations and polychaete worm tissue concentrations, using 2,3,7,8-TCDD as an example COPEC. There was a strongly significant relationship between sediment and polychaete worm tissue 2,3,7,8-TCDD concentrations ((r^2 = 0.91, p=0.0002; Figure 8-2). The regression equation from this analysis was used to predict polychaete worm 2,3,7,8-TCDD concentrations from sediment concentrations. The predicted polychaete worm tissue concentrations were then used in the food web models used to calculate NBSA-wide and assessment zone risk HQs.

There was no change in 2,3,7,8-TCDD HQs for spotted sandpiper or lesser scaup when using the actual polychaete worm tissue data or the predicted worm tissue data derived from the relationship between sediment and worm tissue concentrations. ~~Summary tables of HQ results~~The results of this sensitivity analysis are summarized in Table 8-8. Detailed HQ tables from this sensitivity analysis on the effects on dietary risk of using a sediment to tissue regression to predict worm tissue concentrations are presented in Appendix F.

~~9.3.1.2.3~~8.3.1.2.3 *Treatment of Non-Detects*

The concentrations of summed components (e.g., PCB congeners) that were not detected were assumed to be zero when calculating mixture totals. The effect ~~on HQs~~ of using one-half the ~~detection~~quantitation limit or the full ~~detection~~quantitation limit was evaluated. There were no changes from LOAEL HQ < 1 to HQ ≥ 1 for NBSA-wide and zone HQs regardless of how non-detected values in sums were treated (either as zero, one-half the ~~detection~~quantitation limit, or the full ~~detection~~quantitation limit). ~~Summary tables of HQ~~ The results of this sensitivity analysis are summarized in Table 8-8. Detailed HQ tables from this sensitivity analysis on the treatment of non-detects are presented in Appendix F.

~~9.3.1.2.4~~8.3.1.2.4 *Lesser Scaup Diet*

Although consistent with the USEPA-approved NBSA BERA-recommended exposure factors memorandum (Arcadis 2016), the lesser scaup do not typically include fish in their diet due to beak morphological constraints. The original diet examined in this BERA was 25% blue crab, 25% softshell clam, 25% polychaete worms, and 25% fish (≤ 15 cm). The impact of including fish in the diet of the lesser scaup was evaluated with the following alternative lesser scaup diets in the sensitivity analysis;

- A: 25% crab; 25% clam; 50% worm
- B: 25% crab; 50% clam; 25% worm

- C: 75% clam; 25% worm.

The only effect was that the NBSA-wide Lead LOAEL HQs based on LPR FFS TRVs changed from HQ < 1 to HQ ≥ 1 when lesser scaup diet was assumed to be 75% clam and 25% worm. No other changes to HQs occurred as a result of different lesser scaup diets. The results of this sensitivity analysis are summarized in Table 8-8. Detailed HQ tables from this sensitivity analysis on the lesser scaup diet are presented in Appendix F.

~~9.3.1.2.5~~8.3.1.2.5 *Fish Prey Size*

Fish lengths included in diet composition were based on data from the literature. Only fish < 15 cm were included in the fish dietary components for great blue heron (80% of diet included fish < 15 cm) and double-crested cormorant (100% of diet included fish < 15 cm). These diet assumptions exclude several larger fish samples.

(i) Fish Prey Size – fish < 15 cm and all fish

Heron and double-crested cormorant dietary food web models were evaluated using all fish samples to assess this uncertainty~~. The NBSA-wide~~ and ~~Southeast Zone LOAEL HQs did not change after including all fish in the diet. For the North and Southwest Zones,~~ the following changes ~~from HQs < 1 to HQs ≥ 1~~ occurred:

- For great blue heron:
  - NBSA-wide – no changes
  - The North Zone LPR FFS LOAEL HQs for ~~methylmercury and~~ total PCB ~~Dioxin~~ TEQ ~~-~~bird~~-~~, total dioxin/furan/PCB TEQ bird, mercury, and methylmercury changed from < 1 to ≥ 1 after including all fish in the diet.
  - ~~Southwest~~The North Zone LPR FFS LOAEL ~~HQ~~HQs for ~~total PCB dioxin TEQ bird,~~ mercury changed from ~~≥~~< 1 to ~~<~~≥ 1 after including all fish in the diet.
  - Southwest Zone - no changes
- For double-crested cormorant:
  - ~~North Zone~~NBSA-wide LPR FFS LOAEL HQ for mercury and methylmercury ~~and total PCB TEQ - bird~~ changed from < 1 to ≥ 1 after including all fish in the diet.
  - North Zone LPR FFS LOAEL HQ for methylmercury and Total PCB TEQ Bird changed from < 1 to ≥ 1 after including all fish in the diet.
  - Southeast – no changes
  - Southwest Zone LPR FFS LOAEL HQs for copper, mercury, and methylmercury changed from < 1 to ≥ 1 after including all fish in the diet. LPR FFS LOAEL HQs for total PCB TEQ -and total dioxin/furan/PCB TEQ changed from ≥ 1 to < 1 after including all fish in the diet.

The calculated EPCs for fish of all lengths used and the summary tables of HQ results from this sensitivity analysis on the effect of including fish of all lengths in the bird dietary risk evaluation are presented in Appendix F.

(i) Fish Prey Size – fish < 15 cm and fish > 30 cm

Heron dietary food web models were evaluated using fish > 30 cm to assess the effects of larger fish on dietary risk. The original diet examined in this BERA was 10% blue crab, 10% softshell clam, and 80% fish (≤ 15 cm). The impact of including fish in the diet of the great blue heron was evaluated with the following alternative great blue heron diets in the sensitivity analysis;

- A: 100% fish > 30 cm
- B: 10% clam; 10% crab; 75% fish ≤ 15 cm; 5% fish > 30 cm
- C: 10% clam; 10% clam; 70% fish ≤ 15 cm; 10% fish > 30 cm.

When the great blue heron diet was assumed to consist of 100% fish > 30 cm, mercury and methylmercury LPR FFS LOAEL HQs increased from < 1 to ≥ 1. There were no changes from LOAEL HQ < 1 to HQ ≥ 1 for NBSA-wide and zone HQs for ~~the different great blue heron~~ diets ~~examined~~B or C.

The results of this sensitivity analysis are summarized in Table 8-8.

~~9.3.1.2.6~~8.3.1.2.6 *Other Uncertainties*

Additional uncertainties were identified that are not expected to substantially impact the risk estimates. These uncertainties were identified but not quantitatively evaluated:

- Use of TEFs for calculation of TEQ concentrations in dietary models – Avian TEFs are calculated based on egg injection studies or *in vitro* exposures where the exposure conditions are not likely to be representative (in terms of the suite of congeners and their relative concentrations) of the *in vivo* exposures that result from dietary uptake of dioxins and furans (Van den berg et al. 1998). Therefore, dioxin TEQs calculated using bird TEFs are relevant only to tissue data in which dioxin/furan congeners were measured. There is significant uncertainty in the application of bird TEFs to soil data in the dietary models and dioxin TEQ HQs for these receptors may not be predictive of actual risk.
- Modeled diet – Dietary items were limited to prey species with tissue chemistry data available from the NBSA, including bioaccumulation worm, blue crab, softshell clam, and fish tissue, and did not include other prey items that may be additional components of bird diets, such as zooplankton, squid, shrimp, mammals, amphibians, or aquatic and terrestrial plants. In addition to the limited types of prey used in the bird dietary model, the selection of explicit prey portions may not reflect the feeding behavior of some bird species. Therefore, the representativeness of the dietary estimates (based on available prey tissue data) for actual NBSA bird diets is uncertain.
- Food ingestion rate – food ingestion rates for three of the bird receptors were estimated as a function of body weight using equations from Nagy (2001). The food ingestion rate for great blue heron was based on a model developed by Kushlan (1978) that was also a function of body weight. It is unknown whether the modeled food ingestion rates reflect actual ingestion rates of NBSA birds. The studies used to select ingestion rates have derived these rates using empirical data on birds. The use

of these ingestion rate equations for birds is standard practice in Superfund risk assessments, and it is not likely that ingestion rates vary substantially in a manner that would affect risk outcomes.

- Sediment ingestion rate - Measured incidental sediment ingestion rates used for the spotted sandpiper were based on several other sandpiper species. Measured sediment ingestion rates were available for lesser scaup. Sediment ingestion rates for the other species were based on what is known about each species' feeding habitats and best professional judgment. It is unknown whether the estimated sediment ingestion rates reflect actual ingestion rates of NBSA bird receptors.

~~9.3.2~~8.3.2 Bird Egg Tissue Risk Characterization Uncertainties

This section discusses uncertainties associated with EPCs and selected TRVs that could affect HQ calculations for bird eggs. The uncertainty analysis presented in this section includes qualitative discussions and, in some cases, quantitative evaluations intended to convey the magnitude and direction of uncertainty in the risk estimates. Although many of the generic sources of uncertainty discussed above in the dietary assessment for birds also apply to the bird egg assessment, there are some important differences, including:

- Tissue EPCs – The use of egg tissue eliminates many uncertainties associated with estimating exposure doses, such as uncertainties in the diet composition, SUF, and food and sediment ingestion rates because COPEC concentrations are measured directly in egg tissue.
- Dioxin/furan TEFs – Bird TEFs are calculated based on egg injection studies where the exposure conditions are not likely to be representative (in terms of the suite of congeners and their relative concentrations) of the *in vivo* exposures that result from dietary uptake of dioxins and furans (Van den berg et al. 1998). Therefore, dioxin TEQs calculated using bird TEFs are relevant only to tissue data in which dioxin/furan congeners were measured. Thus, the egg TEQ concentrations have lower uncertainty than the TEQ exposure doses predicted in the dietary assessment for birds.
- Use of egg COPEC and reproductive success data from Parsons (2003) for assessment – While there is uncertainty in relying on a single field study for risk assessment due to the potential for multiple contaminant exposure and the effects of site foraging fidelity behavior by cormorants on exposure, this particular study is important because it: 1) focused on an upper trophic level bird that is likely most exposed to COPECs in the NBSA; 2) was robust in terms of the data collected, inclusions of multiple study sites and reference conditions, number of nest monitored; and 3) was definitive in terms of its findings with respect to reproductive success of the monitored nests. Given that the endpoints measured are of direct important to the AE for birds for this BERA, the uncertainties associated with its use for risk characterization are considered low.

Uncertainties specific to the egg tissue risk estimates for birds in the NBSA BERA are discussed below.

~~9.3.2.1~~8.3.2.1 Egg Tissue TRVs

Similar to the dietary assessment, a key uncertainty associated with the avian egg risk characterization is the selection of TRVs used in the risk calculations.

~~As described above~~Consistent with the approach used to select dietary TRVs discussed in Section 8.3.1.1, the egg tissue TRVs derived for this BERA were selected to overestimate rather than

underestimate risk. As a result, this section focuses on the TRV uncertainty for those COPECs where egg LOAEL HQs ≥ 1, which was the case only for ~~total DDx~~mercury and methylmercury (LPR FFS TRVs only), total DDx, total PCBs, total PCB TEQ bird, and total PCB/dioxin TEQ bird using both sets of TRVs. Each of the NBSA TRVs was the same as was used in the LPRSA BERA (Windward 2019). General uncertainties associated with TRVs are discussed in Appendix D.

For ~~dioxins/furans and~~ TEQ, the NBSA egg LOAEL TRV is based on the 5th percentile from an SSD containing egg-injection studies with five bird species, some of which have been observed in the NBSA (double-crested cormorant, great blue heron). Double-crested cormorant and great blue heron are among the least sensitive birds in the dataset. Due to the limited dataset size, the 5th percentile value is below any empirically measured LOAELs and is a conservative estimate of avian toxicity for TEQ. The LPR FFS LOAEL TRV is also based on the 5th percentile SSD value for a dataset containing chicken toxicity data, which has uncertain relevance for wild species in the NBSA. Both the NBSA and LPR FFS TRVs are similar in magnitude and are based on either limited or conservative datasets, and therefore are assumed to have a similar degree of uncertainty. Importantly, both sets of TRVs likely overestimate risk to double-crested cormorants, as the species-specific LOAEL (4000 ng/kg ww; Powell et al. (1997); Figure 8-6 of LPRSA BERA) is more than an order of magnitude greater than the selected TRVs. Using the double-crested cormorant LOAEL TRV, egg HQs are all well below 1.

Table 8-9. Comparison of NBSA-wide Cormorant Egg Tissue HQs Using Species-Specific TRVs

Constituent	Units	NBSA TRV HQ (0.00025 mg/kg) Windward (2019) SSD 5 species	LPR FFS TRV HQ (0.~~000~~00015 mg/kg) USEPA (2003) Various species	Species-specific TRV HQ (0.004 mg/kg) Powell et al (1997) Cormorant
Dioxins/Furans				
2,3,7,8-TCDD	unitless	0.4	0.6	0.02
Total Dioxin/Furan TEQ Bird	unitless	0.5	0.9	0.03
PCBs				
Total PCB TEQ Bird	unitless	**2**	**3**	0.1
Total Dioxin/Furan/ PCB TEQ Bird	unitless	**2**	**4**	0.1

For total PCBs, the NBSA egg LOAEL TRV is based on the lowest available LOAEL. Although only two studies were available, the selected LOAEL is based on a wild species (ringed turtle dove) and based on reduced hatchability in a two-generation study. Multiple NOAELs above the selected TRVs are available.

The LPR FFS LOAEL TRV is based on an interpolated 25% effect level for reduced hatchability in domestic chicken, which has uncertainty for wild populations. Due to the chronic study duration and use of reproductive data for wild species, the NBSA TRV is considered to be somewhat more predictive of toxicity to wild birds in the NBSA.

For total DDx, the NBSA egg LOAEL TRV is based on the 5th percentile from an SSD containing data for seven bird species, some of which have been observed in the NBSA (kestrel, black duck, and mallard). The LPR FFS egg LOAEL TRV is similar in magnitude but is based on field residue data for brown pelican that were not dose-responsive for the selected endpoint (nesting success). As noted in Appendix D, the use of field-collected egg data created uncertainty in establishing a LOAEL, given the other factors in the field that could potentially influence to reproductive success (e.g., other contaminants and non-chemical stressors).

~~9.3.2.2~~8.3.2.2 Exposure Assumptions

As noted above, the use of measured egg tissue COPEC concentrations eliminates many uncertainties associated with exposure in the bird egg risk assessment. Remaining uncertainties for the bird egg risk assessment were identified and discussed below, but were not quantitatively evaluated:

- Treatment of non-detects for EPCs – All egg tissue concentration data were derived from Parsons (2003), which treated non-detects as zero. Therefore, it was not possible to assess the effects of treating non-detects as half or the full ~~detection~~quantitation limit on bird egg HQs. Based on the results of the non-detect evaluation for the bird dietary assessment, treatment of non-detects as half or full ~~detection~~quantitation limits is unlikely to substantially change risk estimates for bird eggs.
- Sample size – For most COPECs measured by Parsons (2003) in cormorant eggs, sample size was adequate for calculation of tissue EPCs based on 95% UCLs. For methylmercury however, data were available for only three eggs and therefore the maximum detected concentration was selected as the EPC. Because methylmercury concentrations in the three eggs were similar (range 175-320 nanograms per gram [ng/g] ww) and because total mercury concentrations were also not highly variable (range 85 -1130 ng/g ww) in a larger number of samples (N=14), the risk estimates are not expected to be substantially over- or underestimated for methylmercury.

~~9.4~~8.4 Weight of Evidence Risk Findings for Birds

As summarized in Section 3.5, a WOE approach was used to evaluate COPECs with LOAEL HQs ≥ 1 (preliminary COCs; Table 8-10) and potential for unacceptable risk to bird populations in the NBSA.

Table 8-10. Summary of Preliminary COCs for Bird Dietary Assessment

Preliminary COCs	Exposure Area	Range of LOAEL HQs									
		Bird Diet LOE								Bird Egg LOE	
		NBSA-wide TRVs				LPR FFS TRVs				NBSA TRVs	LPR FFS TRVs
		Spotted Sandpiper	Great Blue Heron	Double-crested Cormorant	Lesser Scaup	Spotted Sandpiper	Great Blue Heron	Double-crested Cormorant	Lesser Scaup	Double-crested Cormorant	Double-crested Cormorant
Total PCBs	NBSA-wide	<1	<1	<1	<1	<1	<1	<1	<1	1	15
	Zones	<1	<1	<1	<1	<1	<1	<1	<1	--	--
Total PCB TEQ Bird	NBSA-wide	<1	<1	<1	<1	<1	<1	<1	<1	2	3
	Zones	<1	<1	<1	<1	<1	<1	<1 - 1	<1	--	--
Total Dioxin/Furan/PCB TEQ Bird	NBSA-wide	<1	<1	<1	<1	<1	<1	1	<1	2	4
	Zones	<1	<1	<1	<1	<1	<1	1	<1	--	--
Total DDx (2,4 & 4,4)	NBSA-wide	<1	<1	<1	<1	<1	1	2	<1	2	2
	Zones	<1	<1	<1	<1	<1	1 - 2	1 - 3	<1	--	--
Lead	NBSA-wide	<1	<1	<1	<1	1	<1	<1	<1	--	--
	Zones	<1	<1	<1	<1	<1 - 3	<1	<1	<1 - 1	--	--
Mercury	NBSA-wide	<1	<1	<1	<1	<1	<1	<1	<1	<1	5
	Zones	<1	<1	<1	<1	<1 - 1	<1	<1	<1	--	--
Methylmercury	NBSA-wide	<1	<1	<1	<1	<1	<1	<1	<1	<1	3
	Zones	<1	<1	<1	<1	<1	<1	<1 - 1	<1	--	--

Notes:
-- = not evaluated

For birds, a range of dietary and tissue HQs~~,~~ (including HQs based on sensitivity analyses), as well as a detailed field study of reproductive success of birds that nested and hatched on Shooters Island, provides the available LOEs used in this BERA. These include:

1. HQs based on dietary doses in wildlife food web models estimated using measured prey tissue concentrations compared to literature-derived dietary TRVs specific to the NBSA
2. HQs based on dietary doses in wildlife food web models estimated using measured prey tissue concentrations compared to literature-derived dietary TRVs derived for the LPR FFS (USEPA 2014)
3. HQs based on tissue concentrations measured in double-crested cormorant eggs collected in the NBSA compared to bird-egg based tissue TRVs from the literature
4. Direct measures of reproductive success of double-crested cormorant eggs, hatchlings, and fledglings on Shooters Island.

Dietary HQs were calculated for the NBSA (in the mudflats for spotted sandpiper and great blue heron; NBSA-wide for double-crested cormorant and lesser scaup), and also independently for the north, southeast, and southwest assessment zones. A variety of sensitivity analyses were conducted for the dietary HQs. Egg tissue HQs were estimated for bioaccumulative compounds for the NBSA-wide area.

The HQ assessment is contrasted with the actual measures of reproductive effects from Parsons (2003). This study provides actual data demonstrating a lack of reproductive effects in double-crested cormorants, one of the most exposed upper trophic level birds to COPECs in the NBSA; although, egg predation levels were high at two of the three islands examined. As such, it is given high weight in the risk findings.

Uncertainties in the dietary HQs (Section 8.3.1) and egg-tissue HQs (Section 8.3.2) were evaluated in this WOE assessment according to the factors presented in Section 3.5 to arrive at an overall risk conclusion for each preliminary COC.

~~Avian~~As discussed above in Sections 8.3.1 and 8.3.2, avian TRVs are a key uncertainty in the risk estimates and apply to all avian receptors evaluated in this BERA. ~~For preliminary COCs, Table 8-10 indicates which TRVs have highest reliability for risk estimation in the NBSA BERA based on the uncertainties presented in Section 8.3.1 for dietary TRVs, and Section 8.3.2 for egg tissue TRVs. Avian TRV reliability was considered in the WOE evaluation presented in this section.~~For both dietary and egg tissue TRVs, two sets of TRVs (i.e., NBSA and LPR FFS TRVs) were included for bounding purposes and given equal weight in this WOE evaluation. Additional LOE are discussed below for each receptor and preliminary COC.

~~Table 8-10. Assessment of Comparative TRV Reliability for Risk Estimation~~

~~Preliminary COCs~~	~~Reliability for Risk Estimation in the NBSA BERA~~			
	~~Dietary TRVs~~	~~Dietary TRVs~~	~~Egg TRVs~~	~~Egg TRVs~~
	~~NBSA TRVs~~	~~LPR FFS TRVs~~	~~NBSA TRVs~~	~~LPR FFS TRVs~~
~~Dioxins/Furans & TEQ[a]~~	~~**Moderate** Lowest LOAEL[h]~~	~~**Low** Lowest LOAEL & UF for interspecies sensitivity[d]~~	~~**Moderate** 5th percentile SSD based on limited dataset[f,h]~~	~~**Moderate** 5th percentile SSD based on conservative[e] dataset~~
~~Total PCBs~~	~~**Moderate** Lowest LOAEL[h]~~	~~**Low** Interpolated Value below LOAEL[g]~~	~~**Moderate** Lowest LOAEL[h]~~	~~**Low** Interpolated Value below LOAEL[g]~~
~~Total DDx~~	~~**High** 5th percentile SSD[f,h]~~	~~**Low** LOAEL from field study[b]~~	~~**High** 5th percentile SSD~~	~~**Low** LOAEL from field study[b]~~
~~Lead~~	~~**Moderate** Lowest LOAEL[h]~~	~~**Low** Lowest LOAEL[c]~~	~~NA~~	~~NA~~
~~Mercury & Methylmercury~~	~~**High** 5th percentile SSD[h]~~	~~**Low** Lowest LOAEL & UF for interspecies sensitivity~~	~~**Low** LOAEL for single species[h]~~	~~NA~~

~~**Notes:**~~

~~a. TEQ related COPECs include 2,3,7,8-TCDD, total dioxin TEQ bird, total PCB TEQ bird, total PCB dioxin TEQ bird~~

~~b. This LOAEL is based on field investigation of pelican productivity/reproductive success.~~

~~c. This LOAEL is based on egg production in a domestic species; endpoint relevance uncertain for wild species.~~

~~d. Wild species with AhR Type 1 (high sensitivity) are not known to be present in the NBSA.~~

~~e. Dataset includes reproduction data (egg production) for domestic species and data for sensitive birds (Type I AhR receptor), which is unlikely to represent toxicity wild birds in the NBSA.~~

~~f. SSD includes multiple species resident in the NBSA.~~

~~g. Based on domestic chicken reproductive endpoints, considered uncertain for wild species.~~

~~h. This value was also selected in the LPRSA BERA (Windward 2019).~~

~~NA – Not Applicable~~

~~9.4.1~~8.4.1 Spotted Sandpiper

For spotted sandpiper, dietary LOAEL HQs ≥ 1 were predicted for the COPECs identified in Table 8-11. Sensitivity analyses conducted for this receptor included evaluation of NDs, dietary HQs, use of an uptake model for polychaete worm dietary fraction, and evaluation of an SUF of 1.

Table 8-11. Exceedances of TRVs for Spotted Sandpiper Dietary Assessment

Preliminary COCs	Dietary HQs ≥ 1	
	NBSA TRVs	LPR FFS TRVs
Primary Exposure and Toxicity Assumptions (Table 8-4)		
Lead	None	NBSA-wide; SW zone
Mercury	None	SW zone
Sensitivity Analyses (Table 8-8)		
Total Dioxin/Furan/PCB TEQ Bird	None	SW zone (SUF = 1); NBSA-wide; N and SE zones (ND=QL)
Total HMW PAH	None	N zone (ND=QL)
Lead	None	NBSA-wide; SW zone (SUF =1)
Mercury	None	NBSA-wide; N, SE, and SW zones (SUF =1)

Notes:
ND = not detected; QL = quantitation limit; SUF = site use factor
Sensitivity analysis resulting in LOAEL HQs ≥ 1 is indicated in parentheses; see Section 8.3 for more detail.

No dietary LOAEL HQs ≥ 1 were predicted for any COPECs using the NBSA-selected TRVs in the NBSA-wide area or individual assessment zones~~, or for either set~~ in any of ~~dietary TRVs in~~ the ~~north~~scenarios evaluated in this BERA. Using the primary exposure and ~~southeast assessment zones. Dietary~~toxicity assumptions, dietary LOAEL HQs ≥ 1 were predicted only for lead and mercury using the LPR FFS TRVs in the NBSA-wide (lead only) and Southwest Zone. Sensitivity analyses, including evaluation of a SUF of 1 and calculation of total concentrations using the full quantitation limit for NDs, resulted in identification of total dioxin/furan/PCB TEQ and total HMW PAH as additional preliminary COCs based on LOAEL HQs equal to 1 for the NBSA-wide area and/or one or more individual assessment zones.

Inputs to the HQs, including exposure assumptions (SUFs and dietary composition), EPCs, and TRVs, are discussed below as part of the WOE evaluation for this receptor.

SUF: Evidence of spotted sandpipers has been noted in some of the NBSA mudflats during the breeding season. However, spotted sandpiper is migratory and is potentially present in Newark Bay only during this time. Therefore, the SUF of 0.5 is not likely to underestimate exposure~~.~~ and an SUF of 1 is likely to overestimate exposure. Additionally, limited spotted sandpiper habitat is available along the margins of the NBSA due to the highly urbanized shorelines. The mudflat area evaluated in the NBSA is comprised of unvegetated expanses of mud, fine sand, or clay and encompassing roughly 29 acres. Mudflats in the

individual assessment zones range from about 6.5 acres in the Southwest Zone to 19 acres in the North Zone. By comparison, the breeding territory of spotted sandpipers is small (< 2 acres; Hays 1972). Due to the scarcity of adequate breeding habitat within the NBSA, the NBSA-wide area is considered most relevant for evaluation of spotted sandpiper populations.

Diet: In this BERA, spotted sandpipers were assumed to have a diet of 100% polychaete worm, but their actual diet consists of both terrestrial and aquatic invertebrates. As discussed in Section 8.3.1, the assumed diet likely overestimates exposure to this receptor in the dietary food web models.

EPCs: The Southwest Zone ~~lead and mercury~~ sediment EPCs for all preliminary COCs are based on the maximum detected concentration due to the small dataset size for intertidal mudflats in this zone. A maximum EPC is a less robust exposure estimate for this BERA than EPCs based on a 95% UCL.

Based on the various factors considered in the dietary HQ LOE, the resulting risk conclusions for preliminary COCs are as follows:

- Lead: Unacceptable risk is considered unlikely. This conclusion is based on 1) LOAEL HQs < 1 based on NBSA TRV; 2) low magnitude of the LOAEL HQs ≥ 1 based on the LPR FFS TRV; and 3) conservative assumptions used in the risk estimates (e.g., maximum EPC, exposure assumptions, and TRVs).
- Mercury: Unacceptable risk is considered unlikely. This conclusion is based on 1) LOAEL HQs < 1 based on NBSA TRV; 2) low magnitude of the LOAEL HQs ≥ 1 based on the LPR FFS TRV; and 3) conservative assumptions used in the risk estimates (e.g., maximum EPC, exposure assumptions, and TRVs).
- Total Dioxin/Furan/PCB TEQ: Unacceptable risk is considered unlikely. This conclusion is based on 1) LOAEL HQs < 1 based on NBSA TRV; 2) low magnitude of the LOAEL HQs =(but not greater than) 1 based on the LPR FFS TRV; and 3) conservative assumptions used in the risk estimates (e.g., maximum EPC, exposure assumptions, and TRVs).
- Total HMW PAH: Unacceptable risk is considered unlikely. This conclusion is based on 1) LOAEL HQs < 1 based on NBSA TRV; 2) low magnitude of the LOAEL HQs = (but not greater than) 1 based on the LPR FFS TRV; and 3) conservative assumptions used in the risk estimates (e.g., maximum EPC, exposure assumptions, and TRVs).

~~9.4.2~~8.4.2 Great Blue Heron

For great blue heron, dietary LOAEL HQs ≥ 1 were predicted for the COPECs identified in Table 8-12. Sensitivity analyses conducted for this receptor included evaluation of NDs, dietary HQs, and modified prey size and composition assumptions for the dietary HQs.

Table 8-12. Exceedances of TRVs for Great Blue Heron Dietary Assessment

Preliminary COCs	Dietary HQs ≥ 1	
	NBSA TRVs	LPR FFS TRVs
Primary Exposure and Toxicity Assumptions (Table 8-4)		
Total DDx	None	NBSA-wide; N, SE, & SW zones
Sensitivity Analyses (Table 8-8)		
Total PCB TEQ	None	N zone (Diet = fish all sizes)
Total Dioxin/Furan/PCB TEQ	None	N zone (Diet = fish all sizes)
Total DDx	None	NBSA-wide; N, SE, & SW zones (Diet = fish all sizes)
Mercury	None	N zone (Diet = fish all sizes) NBSA-wide; N, SE, & SW zones (Diet A [100% fish > 30 cm])
Methylmercury	None	NBSA-wide; N, SE, & SW zones (Diet A [100% fish > 30 cm])

No dietary LOAEL HQs ≥ 1 were predicted for any COPECs using the NBSA TRVs in the NBSA-wide area or individual assessment zones~~. Dietary~~ in any of the scenarios evaluated in this BERA. Using the primary exposure and toxicity assumptions, dietary LOAEL HQs ≥ 1 were predicted only for total DDx using the LPR FFS TRVs in the NBSA-wide area and all assessment zones. Sensitivity analyses related to evaluation of alternate diets resulted in identification of total PCB TEQ, total dioxin/furan/PCB TEQ, mercury, and methylmercury as additional preliminary COCs based on LOAEL HQs ≥ 1 for the NBSA-wide area and/or one or more individual assessment zones.

Inputs to the HQs, including SUFs, EPCs, and TRVs, are discussed below as part of the WOE evaluation for this receptor.

SUF: Great blue heron are expected in Newark Bay year-round and this receptor has been observed within the NBSA. Kearny marsh (north assessment zone) seasonally becomes a roost for large numbers of herons and egrets, including great blue heron. This receptor was evaluated using a SUF of 1.0. The SUF likely overestimates great blue heron exposure to the NBSA based on their large foraging range (forage 3 to 7 kilometers from breeding colony; Vennesland and Butler 2011) relative to the area of mudflat habitat within the NBSA (29 acres). Due to the limited areas of mudflats and adequate breeding habitat within the NBSA, the NBSA-wide mudflat area (29 acres) is considered most relevant for evaluation of great blue heron populations.

Diet: Great blue heron were assumed to eat a mixed diet containing primarily fish and also aquatic invertebrates. The primary assumed diet consists of fish in the size range of <15 cm as part of the diet and is not assumed to significantly over- or underestimate exposure for this receptor. The sensitivity evaluations assume that heron eat fish of all sizes as part of a mixed diet or only eat fish less than 30 cm, neither of which are less realistic for this species.

EPCs: In general, EPCs were based on 95% UCL concentrations which are appropriate exposure estimates for a BERA. A maximum EPC is a less robust exposure estimate for this BERA than EPCs

based on a 95% UCL. In cases where a COPEC is infrequently detected, use of the maximum EPC potentially overestimates exposure. These cases include:

- ~~Total DDx in sediment and clam tissue~~Sediment EPCs: Sediment Southwest and Southeast Zone sediment EPCs are based on maximum detected concentrations for all preliminary COCs due to the small dataset size.
- Clam EPCs: The clam dietary fraction is also based on the maximum EPC ~~in these~~for most preliminary COCs in the Southeast and Southwest zones due to small sample size (N < 6) in the Southeast Zone and low FOD in the Southwest Zone.

Based on the various factors considered in the dietary HQ LOE, the resulting risk conclusions for preliminary COCs are as follows:

- Total DDx: Unacceptable risk is considered unlikely. These conclusions are based on 1) LOAEL HQs < 1 using NBSA TRVs; 2) low magnitude of the LOAEL HQs ≥ 1 using LPR FFS TRVs; and 3) conservative assumptions used in the risk estimates (e.g., maximum EPCs in some assessment zones, conservative TRVs, SUFs).
- Total PCB TEQ and total dioxin/furan/PCB TEQ: Unacceptable risk is considered unlikely. These conclusions are based on 1) LOAEL HQs < 1 using NBSA TRVs; 2) low magnitude of the LOAEL HQs ≥ 1 using LPR FFS TRVs; and 3) conservative assumptions used in the risk estimates (e.g., maximum EPCs in some assessment zones, conservative TRVs, SUFs).
- Mercury and methylmercury: Unacceptable risk is considered unlikely. These conclusions are based on 1) LOAEL HQs < 1 using NBSA TRVs; 2) low magnitude of the LOAEL HQs ≥ 1 using LPR FFS TRVs; and 3) conservative assumptions used in the risk estimates (e.g., maximum EPCs in some assessment zones, conservative TRVs, SUFs).

~~9.4.3~~8.4.3 Double-crested cormorant

For double-crested cormorant, LOAEL HQs ≥ 1 were predicted for the COPECs identified in Table 8-13. Sensitivity analyses conducted for this receptor included evaluation of NDs for dietary HQs, modified prey size and composition assumptions for the dietary HQs, and use of a species-specific TRV for the egg tissue HQs.

Table 8-13. Exceedances of TRVs for Double-Crested Cormorant Dietary Assessment

Preliminary COCs	Dietary LOAEL HQs ≥ 1		Egg Tissue LOAEL HQs ≥ 1	
	NBSA TRVs	LPR FFS TRVs	NBSA TRVs	LPR FFS TRVs
Primary Exposure and Toxicity Assumptions (Table 8-4)				
Total Dioxin/Furan/PCB TEQ Bird	None	NBSA-wide; N, SE, & SW zones	NBSA-wide	NBSA-wide
Total PCB TEQ Bird	None	SW zone	NBSA-wide	NBSA-wide
Total PCBs	None	None	NBSA-wide	NBSA-wide
Total DDx	None	NBSA-wide; N, SE, & SW zones	NBSA-wide~~None~~	NBSA-wide
Mercury	None	None	None	NBSA-wide
Methylmercury	None	SE zone	None	~~NC~~NBSA-wide
Sensitivity Analyses (Table 8-8)				
Total PCB TEQ Bird	None	N zone (Diet = fish all sizes)	None	None
Total Dioxin/Furan/PCB TEQ Bird	None	NBSA-wide; N, SE, & SW zones (Diet = fish all sizes)	None	None
Total DDx	None	NBSA-wide; N, SE, & SW zones (Diet = fish all sizes)	NA	NA
Copper	None	SW zone (Diet = fish all sizes)	NA	NA
Mercury	None	NBSA-wide, N, SE, SW zones (Diet = fish all sizes)	NA	NA
Methylmercury	None	NBSA-wide, N, SE SW zones (Diet = fish all sizes)	NA	NA

No dietary LOAEL HQs ≥ 1 were predicted for any COPECs using the NBSA TRVs in the NBSA-wide area or individual assessment zones~~. Dietary~~ in any of the scenarios evaluated in this BERA. Using the primary exposure and toxicity assumptions, dietary LOAEL HQs ≥ 1 were predicted for total dioxin/furan/PCB TEQ and total DDx using the LPR FFS TRVs in the NBSA-wide area and all assessment zones, total PCB TEQ in the Southwest Zone, and methylmercury in the Southeast Zone. Sensitivity analyses related to evaluation of alternate diets resulted in identification of copper as additional

preliminary COCs based on dietary LOAEL HQs ≥ 1 for the NBSA-wide area and/or one or more individual assessment zones.

Egg tissue LOAEL HQs ≥ 1 were predicted for total dioxin/furan/PCB TEQ, ~~Total~~total PCB TEQ, ~~and total~~Total PCBs ~~in the NBSA-wide area using both sets of TRVs, and~~ , and total DDx in the NBSA-wide area using both sets of TRVs, and mercury and methylmercury in the NBSA-wide area using LPR FFS TRVs only. Quantitative sensitivity analyses were not conducted for egg tissue HQs.

Inputs to the dietary and tissue HQs, including exposure assumptions (SUFs and dietary composition), EPCs, and TRVs where appropriate, are discussed below as part of the WOE evaluation for this receptor.

SUF: Double-crested cormorant are expected in Newark Bay year-round and this receptor has been observed within the NBSA. Nests are seasonally present on Shooters Island (Southwest Zone) and have been observed on in water features like buoys, markers, and other structures (see Appendix B). This receptor was evaluated using a SUF of 1.0. Based on a foraging range of roughly 5 to 10 miles from the nest colony (Palmer 1962), the selected SUF is not anticipated to over- or underestimate exposure for this receptor. Based on foraging range, the NBSA-wide area is considered most reliable for evaluating exposure estimates for double-crested cormorants.

Diet: Double-crested cormorant were assumed to eat a diet consisting exclusively of fish less than 15 cm~~. The~~ in the primary exposure assumptions. The primary assumed diet is not expected to significantly over- or underestimate exposure for this receptor. The sensitivity evaluation assumes that cormorant eat fish of all sizes, which is less realistic for this species.

EPCs: In general, EPCs were based on 95% UCL concentrations which are appropriate exposure estimates for a BERA. A maximum EPC is a less robust exposure estimate for this BERA than EPCs based on a 95% UCL. In cases where a COPEC is infrequently detected, use of the maximum EPC potentially overestimates exposure. EPC factors considered in the WOE include:

- Total dioxin~~-~~/furan/PCB TEQ bird: In egg tissue, PCBs account for about 90% of the total Dioxin/Furan/PCB ~~dioxin~~ TEQ concentration.
- Methylmercury in bird eggs: Bird egg EPCs are based on maximum detected concentrations in the NBSA-wide area calculated from eggs analyzed for methylmercury (N=3). Bird eggs, collected on Shooters Island, are considered representative of the NBSA-wide area and not evaluated by individual assessment zones.

TEFs: As discussed in Section 8.3.1, avian TEFs are calculated based on egg injection studies. Dioxin TEQs calculated using bird TEFs are relevant only to tissue data in which dioxin/furan congeners were measured. There is significant uncertainty in the application of bird TEFs to soil data in the dietary models. Use of egg tissue data eliminates this uncertainty.

Based on the various factors considered in the dietary and egg tissue HQ LOEs, the resulting risk conclusions for preliminary COCs are as follows:

- Total ~~PCB~~ dioxin/furan/PCB TEQ bird and total PCB TEQ: Unacceptable risk is considered unlikely. Although LOAEL HQs ≥ 1 were predicted using both sets of tissue TRVs and tissue HQs were considered to have the least uncertainty for the HQ LOE, the HQs likely overestimate toxicity to this species based on the use of LOAEL TRVs based on 5th percentile values from SSDs. The species-specific egg LOAEL is more than 10-fold greater than the selected LOAEL TRVs, which is consistent

with genetic sequencing data indicating a low sensitivity AhR receptor (Type III; Farmahin et al. 2012) for this species. Additionally, Parsons (2003) found no relationship between reproductive success (e.g., egg production, hatching success, and chick survival) and COPEC concentrations in eggs and other bird tissues from Shooters Island, located within the NBSA, and two other islands in New York Harbor and Long Island. This site-specific measure of toxicity provides an important LOE indicating that COPEC concentrations in the NBSA are not adversely impacting reproductive success in cormorant populations.

- Total PCBs: Unacceptable risk is considered unlikely based on the same LOE as described above for TEQ.
- Total DDx: Unacceptable risk is considered unlikely. The risk conclusions are based on 1) LOAEL HQs < 1 using both tissue and dietary NBSA TRVs; 2) EPCs and SUF unlikely to over- or underestimate exposure; 3) low magnitude of the LOAEL HQs ≥ 1 using ~~conservative~~ LPR FFS TRVs; and 4) no effect on reproductive success found by Parsons (2003).
- ~~Methylmercury~~Mercury and methylmercury: Unacceptable risk is considered unlikely. These conclusions are based on 1) LOAEL HQs < 1 using dietary NBSA TRVs ~~in the NBSA-wide area~~; 2) an SUF unlikely to over- or underestimate exposure; 3) dietary sediment and prey tissue EPCs indicate low variability of mercury and methylmercury concentrations within the NBSA; 4) egg LOAEL HQs < 1 using NBSA TRVs; and 5) low magnitude of the dietary LOAEL HQs ≥ 1 ~~in the Southeast Zone using conservative~~using LPR FFS TRVs. For the HQ LOEs, dietary ~~NBSA~~ HQs were considered to have least uncertainty for predicting risk based on ~~confidence in~~ the ~~TRV and~~ small dataset available for egg tissue EPCs. Site-specific toxicity data from Parsons (2003) support these conclusions.
- Copper: Unacceptable risk is considered unlikely. These conclusions are based on 1) LOAEL HQs < 1 using dietary NBSA TRVs; 2) an SUF unlikely to over- or underestimate exposure; and 5) low magnitude of the dietary LOAEL HQs ≥ 1 in the Southwest Zone sensitivity analysis using LPR FFS TRVs. Site-specific toxicity data from Parsons (2003) support these conclusions.

~~9.4.4~~8.4.4 Lesser Scaup

For lesser scaup, there were no LOAEL HQs ≥ 1 for the NBSA TRVs, and only lead in the Southwest Zone for the LPR FFS TRVs (Table 8-14~~).~~) using the primary exposure and toxicity assumptions. Sensitivity analyses conducted for this receptor included evaluation of ND dietary HQs, modified prey composition for the dietary HQs, and use of an uptake model for polychaete worm dietary fraction. Sensitivity analyses related to evaluation of alternate diets did not result in identification of additional preliminary COCs.

Table 8-14. Exceedances of TRVs for Lesser Scaup Dietary Assessment

Preliminary COCs	Dietary HQs ≥ 1	
	NBSA TRVs	LPR FFS TRVs
Primary Exposure and Toxicity Assumptions (Table 8-4)		
Lead	None	SW zone
Sensitivity Analyses (Table 8-8)		
Lead	None	NBSA-wide and SW zone (Diet C [75% clam; 25% worm])

Inputs to the dietary and tissue HQs, including exposure assumptions (SUFs and dietary composition), EPCs, and TRVs where appropriate, are discussed below as part of the WOE evaluation for this receptor.

SUF: Lesser scaup breeds in the boreal forests of northern North America and is expected in Newark Bay only during the winter or non-breeding season (Anteau et al. 2014) and was therefore given an SUF of 0.5. Home range information was not located for lesser scaup, but observations indicate a relatively large foraging area (see Appendix B). For evaluation of lesser scaup populations, the NBSA-wide area was considered to be most appropriate for developing exposure estimates. Based on the available information, the selected SUF is considered more likely to overestimate rather than underestimate exposure to this receptor.

Diet: The prey diet evaluated in this BERA consists of species with high potential for uptake and is likely representative of the scaup diet, foraged from bottom sediments in shallow water. Sensitivity analyses conducted for varying diets had little effect on the estimated HQs.

EPCs: The Southwest Zone lead EPC is based on the maximum detected concentration due to the small dataset. A maximum EPC is a less robust exposure estimate for this BERA than EPCs based on a 95%UCL.

Based on the various factors considered in the dietary HQ LOE, the resulting risk conclusions for preliminary COCs are as follows:

- Lead: Unacceptable risk is considered unlikely. This conclusion is based on 1) LOAEL HQs < 1 using the NBSA TRVs; 2) low magnitude of the LOAEL HQs ≥ 1 using LPR FFS TRVs; and 3) conservative assumptions used in the risk estimates (e.g., maximum EPCs, conservative TRVs).

~~9.4.5~~8.4.5 Summary of Risk Findings for Birds

Spotted sandpiper, great blue heron, double-crested cormorant, and lesser scaup were evaluated using dietary HQs calculated for the NBSA-wide and individual assessment zones using two sets of TRVs (NBSA and LPR FFS~~).~~) and including a variety of quantitative sensitivity analyses. For double-crested cormorant, egg tissue HQs were also evaluated for the NBSA-wide exposure area using two sets of tissue TRVs when available.

Overall, no unacceptable risk to birds potentially exposed to COPECs in the NBSA is expected. Conclusions for the bird evaluations are based on the WOE evaluation presented above in Section 8.4 and are summarized below:

- For spotted sandpiper (benthivorous bird):
  - ~~Dietary~~Using primary exposure and toxicity assumptions, dietary LOAEL HQs < 1 were estimated for all COPECs using all exposure area and TRVs assumptions, except for lead and mercury using LPR FFS TRVs.
  - Sensitivity analyses, including evaluation of a SUF of 1 and calculation of total concentrations using the full quantitation limit for NDs, resulted in identification of total dioxin/furan/PCB TEQ and total HMW PAH as additional preliminary COCs based on LOAEL HQs equal to 1 for the NBSA-wide area and/or one or more individual assessment zones.
  - Unacceptable risk is considered unlikely for lead ~~and~~, mercury, total dioxin/furan/PCB TEQ, and total HMW PAH based primarily on LOAEL HQs < 1 using NBSA TRVs, the low magnitude of the LPR FFS LOAEL HQs ≥ 1, and the use of exposure assumptions that likely overestimate risk to spotted sandpiper populations. ~~LOAEL HQs ≥ 1 were predicted in the NBSA-wide area and Southwest Zone only.~~
- For great blue heron (piscivorous bird):
  - ~~Dietary~~Using primary exposure and toxicity assumptions, dietary LOAEL HQs < 1 were estimated for all COPECs using all exposure area and TRVs assumptions, except for total DDx and using LPR FFS TRVs.
  - Sensitivity analyses related to evaluation of alternate diets resulted in identification of total PCB TEQ, total dioxin/furan/PCB TEQ, mercury, and methylmercury as additional preliminary COCs based on LOAEL HQs ≥ 1 for the NBSA-wide area and/or one or more individual assessment zones.
  - Unacceptable risk is considered unlikely for total DDx, total PCB TEQ, total dioxin/furan/PCB TEQ, mercury, and methylmercury based primarily on LOAEL HQs < 1 using NBSA TRVs, the low magnitude of the LPR FFS LOAEL HQs ≥ 1, and the use of exposure assumptions that likely overestimate risk to great blue heron populations.
- For double-crested cormorant (piscivorous bird):
  - Using primary exposure and toxicity assumptions, dietary LOAEL HQs < 1 were estimated for all COPECs using all exposure area and TRVs assumptions, except for total dioxin/furan/PCB dioxin TEQ, total PCB TEQ, total PCBs, total DDx, and methylmercury using LPR FFS TRVs only.
  - ~~Unacceptable risk is considered unlikely for total DDx~~Sensitivity analyses related to evaluation of alternate diets resulted in identification of copper and mercury as additional preliminary COCs based ~~primarily~~ on dietary LOAEL HQs ~~<~~≥ 1 ~~using NBSA TRVs, and~~for the ~~low magnitude of the LPR FFS~~NBSA-wide area and/or one or more individual assessment zones.
  - Egg tissue LOAEL HQs ≥ 1~~. Dietary LOAEL HQ ≥ 1 was~~ were predicted for total ~~DDx in all exposure areas using LPR FFS TRVs only.~~
- ~~For double-crested cormorant (piscivorous bird):~~
  - ~~Dietary and egg tissue LOAEL HQs < 1 were estimated for all COPECs using all exposure area and TRVs assumptions, except for total PCB dioxin~~ dioxin/furan/PCB TEQ, total PCB TEQ, ~~and~~

total PCBs ~~(egg tissue only)~~, and total DDx in the NBSA-wide area using both sets of TRVs, ~~total DDx using LPR FFS TRVs only,~~and mercury and methylmercury ~~using LPR FFS TRVs only (dietary model~~ in the ~~southeast assessment zone only).~~NBSA-wide area using LPR FFS TRVs only. Quantitative sensitivity analyses were not conducted for egg tissue HQs.

- Unacceptable risk is considered unlikely for total ~~PCB/~~dioxin/furan/PCB TEQ, total PCB TEQ, total PCBs, total DDx, copper, mercury, and methylmercury.
  - Unacceptable risk is considered unlikely for total ~~PCB~~ dioxin/furan/PCB TEQ, total PCB TEQ, and total PCBs. These conclusions are based on HQs calculated using a species-specific LOAEL TRV and a site-specific reproductive success study (Parsons 2003) that found no effect of COPECs on hatchability, fledging success, and other reproductive parameters in double-crested cormorant. Although LOAEL HQs ≥ 1 were predicted for ~~total PCB dioxin TEQ~~these preliminary COCs in ~~all exposure areas~~some scenarios evaluated in this BERA using NBSA and/or FFS dietary ~~LPR FFS TRVs, the NBSA-wide area using both sets of tissue TRVs, for total PCB TEQ in the southwest assessment zone using dietary LPR FFS TRVs, and the NBSA-wide area using both sets of~~and tissue TRVs, these HQs are likely overestimated.
  - For total DDx, these conclusions are based on LOAEL HQs < 1 using both tissue and dietary NBSA TRVs; EPCs and SUF unlikely to over- or underestimate exposure; and low magnitude of the LOAEL HQs ≥ 1 using ~~conservative LPR FFS TRVs. LOAEL HQs ≥ 1 were predicted in all exposure areas using the LPR FFS dietary and egg tissue~~LPR FFS TRVs.
  - For mercury and methylmercury, these conclusions are based on LOAEL HQs < 1 using dietary and egg tissue NBSA TRVs ~~in the NBSA-wide area~~; an SUF unlikely to over- or underestimate exposure; dietary EPCs indicate low variability of mercury and methylmercury concentrations within the NBSA; and low magnitude of the LOAEL HQs ≥ 1 ~~in the Southeast Zone using conservative~~using LPR FFS TRVs. ~~Dietary~~
  - For copper, these conclusions are based on LOAEL HQs ~~≥ 1 were predicted for methylmercury in the Southeast Zone~~ < 1 using ~~LPR~~NBSA and FFS TRVs and primary exposure assumptions, LOAELs ≥ 1 were identified in the Southwest Zone only in sensitivity analyses with diet and were low in magnitude.
  - For all preliminary COCs, site-specific toxicity data from Parsons (2003) support the conclusion that unacceptable risk is unlikely.

- For lesser scaup (omnivorous bird):
  - ~~Dietary~~Using primary exposure and toxicity assumptions, dietary LOAEL HQs < 1 were estimated for all COPECs using all exposure area and TRVs assumptions, except for lead using LPR FFS TRVs.
  - Sensitivity analyses related to evaluation of alternate diets did not result in identification of additional preliminary COCs.
  - Unacceptable risk is considered unlikely for lead based primarily on LOAEL HQs < 1 using the NBSA TRVs; low magnitude of the LOAEL HQs ≥ 1; and conservative assumptions used in the

risk estimates (e.g., maximum EPCs, conservative TRVs). ~~LOAEL HQs ≥ 1 were predicted only in the Southwest Zone using LPR FFS TRVs.~~

Based on the results of this BERA, potential exposures to most COPECs in the NBSA are not expected to pose unacceptable risk to birds. For birds, exceedances of dietary LOAEL TRVs were found for TEQs and total DDx in double-crested cormorants using only the LPR FFS TRVs, and the HQs for these were at or near 1. For cormorants, egg tissue LOAEL HQs ≥ 1 were also predicted using both sets of tissue TRVs; however, given the low HQs, conservative TRVs relative to cormorant-specific toxicity data for eggs, and lack of evidence on reproductive success from the Parsons (2003) study, risks to cormorants are unlikely. As presented in Table 8-15, the potential for unacceptable risk to bird populations (identified based on LOAEL HQs ≥1 and WOE supporting the conclusion of unacceptable risk) was not identified.

~~10 MAMMAL ASSESSMENT~~

9 MAMMAL ASSESSMENT

Mammals are part of the upper trophic levels of the food web in the NBSA. There are two feeding guilds of mammals evaluated in this BERA: omnivores (consume plant and animal matter), and piscivores (consume fish).

The following are the Assessment ~~Endpoints~~Endpoint and Risk ~~Questions~~Question for mammals that are addressed in this section.

Assessment Endpoint: Survival, growth, and/or reproduction of mammals.

Risk Question 1: *Does the daily dose of COPECs received by mammals (including piscivorous, omnivorous, and invertivorous) from consumption of the tissues of prey species and from other media in the NBSA exceed the TRVs for survival, growth, and/or reproduction of mammals?*

Risk Question 1 is addressed by the mammal dietary risk assessment presented in Section 9.1. The life history profiles for the mammal receptors identified for the tissue and dietary risk assessments are provided in Appendix B.

~~10.1~~9.1 Dietary Assessment

A dietary assessment was conducted for each of the three selected mammal species (river otter, muskrat, and harbor seal). This section describes how exposure and effects concentrations were derived and presents the HQs for the dietary assessment. Section 9.2 presents the uncertainties associated with the dietary assessment for mammals. The COPECs evaluated for the bird dietary assessment are listed in Table 3-5 and include: 2,3,7,8-TCDD, total dioxin/furan TEQ, total PCBs (209 congeners), total PCB TEQ, total dioxin/furan/PCB TEQ, pesticides (total chlordane, total DDx, dieldrin, and hexachlorobenzene), total PAHs, and inorganics/metals (arsenic, cadmium, chromium, copper, lead, mercury/methylmercury, nickel, selenium, silver, and zinc).

~~10.1.1~~9.1.1 Exposure

This section presents the methods for calculating exposure doses, including descriptions of the selection of parameters for the dose equation, composition of prey in diet, exposure areas, and EPCs in prey.

~~10.1.1.1~~9.1.1.1 Dietary Dose Model

Dietary doses for mammals were estimated based on ingestion of biota (i.e., prey) and incidental ingestion of sediment. Surface water ingestion was not included in mammal dietary assessment because mammals were not expected to drink the saltwater of the NBSA. Dietary doses were estimated as milligrams of each COPEC ingested per kilogram of body weight (mg/kg bw-day) per day using the following equation:

ADD_{total} = = $[(EPC_s * SIR/BW * DF_s) + \Sigma(EPC_k * FIR_k/BW * DF_k)] * SUF$ ***Equation 9-1***

where,

ADD_{total} = average daily dose (mg/kg bw/day)

EPC_s = exposure point concentration in sediment (mg/kg, dw)

SIR = ingestion rate for sediment (kg/day, dw)

BW = body weight of organism (kg)

DF_s = dietary fraction of sediment ingested (range 0 to 1)

EPC_k = EPC for the kth food type (mg/kg day, ww)

FIR_k = ingestion rate for the kth food type (kg/day, ww)

DF_k = dietary fraction of intake of the kth food type (range 0 to 1)

SUF = site use factor (unitless); proportion of time the selected species spends foraging in the NBSA

The body weights, ingestion rates, diet composition, and SUFs for river otter, muskrat, and harbor seal are described in Section 9.1.1.2.

~~10.1.1.2~~9.1.1.2 Body Weights, Ingestion Rates, Prey Composition, Site Use Factors, Exposure Areas

Body weights, ingestion rates for sediment and food, SUFs, and diet composition for use in the dietary dose calculations were obtained from the literature as summarized in Table 9-1.

The dietary fraction of each component in each species' diet is based on information obtained from the literature (reviewed in Appendix B) and is presented in Table 9-1. Prey ingested by mammal species were limited to whole-body samples only from prey types for which empirical tissue chemistry data from the NBSA were available (polychaete worm, blue crab, softshell clam, and fish). Fish were separated by length into two groups: fish ≤ 15 cm and fish ≤ 30 cm (Appendix C; Tables C-2 and C-3, respectively)

River otters generally prey on fish that are 2 to 50 cm in length, with an average prey length of approximately 30 cm (Melquist et al. 1981). For the NBSA, river otter diet was limited to fish ≤ 30 cm (comprising 80% of the diet evaluated). However, river otter from some locations have been observed to consume fish up to 50 cm in length (Tumlison et al. 1986; Wise et al. 1981; Melquist et al. 1981) and may infrequently consume fish up to 70 cm in length. Based on the variety of prey river otters may consume, the remaining diet evaluated in the NBSA was made up of 5% polychaete worms, 5% blue crabs, and 10% softshell clams.

Muskrats are herbivorous and eat the basal parts, rhizomes, and leaves of aquatic emergent vegetation (Dozier 1953; Errington 1939). Although they consume mostly plant material, they will also eat some fish, crustaceans, dead birds, and frogs (Errington 1939; Johnson 1925; Willner et al. 1980; cited in USEPA 1993). A diet of 85% vegetation (estimated from sediment using an uptake factor), 5% softshell clams, and 10% fish (≤ 15 cm) was evaluated in the NBSA. The muskrat diet was assumed to consist of 85% aquatic/emergent plants. However, plants were not collected in the NBSA. Instead, soil to plant ~~bioaccumulation factors (~~BAFs~~)~~ derived from USEPA (2007b) were applied to sediment data to estimate

concentrations in aquatic/emergent plant diet (Table 9-2). Soil to plant BAFs were not available for methylmercury and ~~BAFs were~~a BAF equal to 1.0 was assumed ~~to be 1.0~~ for this COPEC.

The diet of harbor seals varies seasonally and includes bottom-dwelling fishes (e.g., flounder), invertebrates (e.g., octopus), and species that can be caught in periodic spawning aggregations (e.g., herring) (Everitt et al. 1981; Lowry and Frost 1981; Pitcher and Calkins 1979; Roffe and Mate 1984; cited in USEPA 1993). Fish species between 4 and 28 cm are consumed, with an average of 6 to 18 cm (Brown and Mate 1983; cited in USEPA 1993). Therefore, fish ≤ 30 cm were included as 90% of the harbor seal diet evaluated in the NBSA. The remaining prey items evaluated as part of the harbor seal diet included 5% blue crab and 5% softshell clam.

While fish, blue crab, and softshell clam data were field collected, the worm tissue data were based on the laboratory bioaccumulation study in which worms were exposed to homogenized sediment collected from the 0 to 15-cm depth horizon.

The SUFs for the representative mammal receptors are provided in Table 9-1. The muskrat is assumed to use the NBSA for the entire year; however, the river otter and harbor seal are transient in the NBSA and assumed to be present for only 50% of the time. According to the USEPA's Wildlife Exposure Handbook (USEPA 1993), river otters inhabit "some" marine environments and are found in "non-polluted waterways" and areas showing "little human impact." Harbor seals are found as far south as eastern Long Island but generally winter offshore. In addition, harbor seals require adequate places to haul out, which are not found in the NBSA. The assumption of a 50% SUF is reasonable based on this information, and professional judgement regarding the life history and available habitat in the NBSA for these species.

Potential risk from prey and sediment ingestion for river otter and harbor seal was evaluated NBSA-wide and by assessment zone (north, southeast, southwest). Muskrat were expected to forage only along shoreline areas with adequate habitat coverage (USEPA 1993); therefore, risk from the aquatic plant portion of their diet (derived from sediment to plant BAFs, discussed above) and incidental sediment ingestion was assessed only from sediment samples collected along the shoreline. The softshell clam and fish portion of the muskrat diet was evaluated NBSA-wide and by assessment zone. River otter included polychaete worm as a portion of their diet; however, polychaete worm sample size was not large enough to be evaluated by assessment zone. Therefore, risk from ingestion of polychaete worms were assessed NBSA-wide in the zone risk calculations for this species. See Tables 4-10 and Figure 4-4 for sediment samples assigned to each assessment zone, mudflats, and shoreline areas, and Table 4-11 for tissue samples assigned to each assessment zone.

~~10.1.1.3~~9.1.1.3 Exposure Point Concentrations

EPCs were calculated for each of the media types (i.e., prey and sediment) to calculate dietary doses using Equation 9-1. For prey concentrations, EPCs were calculated separately for each of the prey types (worms, blue crab, softshell clam, fish ≤ 15 cm, fish ≤ 30 cm) as the UCL for each prey group. EPCs for prey were calculated NBSA-wide and by assessment zone (except polychaete worm, where EPCs were calculated NBSA-wide only due to limited sample size). Plants were not collected in the NBSA; therefore, soil to plant BAFs derived from USEPA (2007c) were applied to sediment data to estimate concentrations in aquatic/emergent plant diet (Table 9-2). Soil-to-plant BAFs were not available for methylmercury; BAFs were assumed to be 1.0 for this COPEC. For sediment, the EPCs were equal to the UCLs using data from the relevant exposure areas for each mammal species. ~~UCL concentrations were calculated using~~

~~USEPA's ProUCL® statistical package (Version 5.1) (USEPA 2015b).~~UCL concentrations were calculated using USEPA's ProUCL® statistical package (Version 5.1) (USEPA 2015b).[11] For each dataset with fewer than six detected samples or fewer than six total samples, a UCL was not calculated; instead, the maximum detected concentration was used as the EPC. If no samples were detected, then the maximum non-detect value was used as the EPC. The UCLs used to calculate mammals dietary EPCs are presented in Appendix C.

~~10.1.1.4~~9.1.1.4 Estimated Doses

Dietary doses were calculated based on Equation 9-1 using the prey and sediment ingestion rates, species body weights, and the prey composition from Table 9-1; and the EPCs from Appendix C. These doses are presented in Appendix E.

~~10.1.2~~9.1.2 Effects

Dietary mammal TRVs for the COPECs identified in Table 3-5 are provided in Table 9-3, and discussed in detail in Appendix D. The selected TRVs were primarily derived from the LPRSA BERA TRV database (Windward 2019~~).~~) and are consistent with the Final LPRSA BERA TRVs for COPECs common to both areas (NBSA and LPRSA). For the LPRSA BERA, Windward (2019) conducted a literature search for relevant toxicological studies, and derived acceptable NOAELs and LOAELs. The lowest acceptable LOAEL and highest NOAEL below the LOAEL from the same study was selected. Several NBSA BERA COPECs were not evaluated in the LPRSA BERA mammal dietary risk assessment (i.e., chromium, silver, total PAHs, ~~2,3,7,8-TCDD,~~ total DDx, total chlordane, and hexachlorobenzene). ~~TRVs for arsenic, cadmium, copper, lead, mercury/methylmercury, nickel, selenium, zinc, total PCBs, total PCB TEQ – mammal, total PCB dioxin/furan TEQ – mammal, total dioxin/furan TEQ – mammal, and dieldrin~~TRVs for these COPECs were available in the LPRSA BERA TRV database (Windward 2019). The selected ~~NBSA~~mammal dietary TRVs for ~~these additional~~ COPECs are further discussed in Appendix D.

In addition, mammal dietary TRVs from the LPR FFS (USEPA 2014) were also assessed and are included in the TRV summary table (Table 9-3). LPR FFS mammal dietary TRVs are available for 2,3,7,8-TCDD, total PCBs, total DDX, dieldrin, HMW PAHs, LMW PAHs, copper, lead, and mercury/methylmercury.

~~10.1.3~~9.1.3 Risk Characterization

This section presents the comparison of mammal dietary doses to NBSA BERA and LPR FFS TRVs to calculate HQs for river otter, muskrat, and harbor seal. NBSA-wide NOAEL and LOAEL HQs for the mammal dietary assessment based on NBSA BERA TRVs are presented in Table 9-4a. Assessment zone (i.e., north, southeast, and southwest) NOAEL and LOAEL HQs based on NBSA BERA TRVs are presented in Tables 9-4b through 9-4d. The HQ calculations for each COPEC and mammal species are provided in Appendix E. A summary of COPECs that exceed their TRVs (i.e., HQs that equal or exceed 1) is provided in Table 9-5.

[11] The UCL recommended by USEPA's ProUCL® software (typically the 95% UCL, but in some cases the 97.5% or even the 99% UCL) was used. The selected UCL statistic is presented in Appendix C-1.

Table 9-5. Summary of TRV Exceedances for Mammal Dietary Assessment

~~Preliminary COCs~~	~~NBSA and LPR FFS HQs ≥ 1.0~~								~~NBSA and LPR FFS HQs ≥ 1.0~~							
	~~River Otter~~								~~Muskrat~~							
	~~NBSA-wide~~		~~North~~		~~SE~~		~~SW~~		~~NBSA-wide~~		~~North~~		~~SE~~		~~SW~~	
	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~
~~Dioxins/Furans~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~2,3,7,8-TCDD~~	-	~~⊖~~	~~⊖~~	~~●~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~
~~Total Dioxin/Furan TEQ Mammal~~	-	~~⊖~~	~~⊖~~	~~●~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~
~~PCBs~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~Total PCB TEQ Mammal~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~
~~Total Dioxin/Furan/ PCB TEQ Mammal~~	-	~~⊖~~	~~⊖~~	~~●~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~
~~Pesticides~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~Hexachlorobenzene~~	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-
~~Inorganics~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
~~Chromium~~	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-
~~Lead~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	~~⊖~~	~~⊖~~
~~Nickel~~	-	-	-	-	-	-	-	-	-	-	-	-	-	-	~~⊖~~	-
~~Silver~~	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-
~~Zinc~~	-	-	-	-	-	-	-	-	~~⊖~~	-	-	-	-	-	~~⊖~~	-

~~Preliminary COCs~~	~~NBSA and LPR FFS HQs ≥ 1.0~~							
	~~Harbor Seal~~							
	~~NBSA-wide~~		~~North~~		~~SE~~		~~SW~~	
	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~	~~NBSA~~	~~LPR FFS~~
~~Dioxins/Furans~~	-	-	-	-	-	-	-	-
~~2,3,7,8-TCDD~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~
~~Total Dioxin/Furan TEQ Mammal~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~
~~PCBs~~	-	-	-	-	-	-	-	-
~~Total PCB TEQ Mammal~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~
~~Total Dioxin/Furan/ PCB TEQ Mammal~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~
~~Pesticides~~	-	-	-	-	-	-	-	-
~~Hexachlorobenzene~~	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-
~~Inorganics~~	-	-	-	-	-	-	-	-
~~Chromium~~	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-
~~Lead~~	-	-	-	-	-	-	-	-
~~Nickel~~	-	-	-	-	-	-	-	-
~~Silver~~	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-	~~⊖~~	-
~~Zinc~~	-	-	-	-	-	-	-	-

Preliminary COCs	NBSA and FFS LOAEL-based HQs ≥ 1.0								NBSA and FFS LOAEL-based HQs ≥ 1.0							
	River Otter								Muskrat							
	NBSA-wide		North		SE		SW		NBSA-wide		North		SE		SW	
	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS
Dioxins/Furans																
2,3,7,8-TCDD		○	○	●		○		○		○		○		○		○
Total Dioxin/Furan TEQ Mammal		○	○	●		○		○		○		○		○		○
Polychlorinated Biphenyls																
Total PCBs																
Total PCB TEQ Mammal		○		○		○		○		○		○		○		○
Total Dioxin/Furan/PCB TEQ Mammal		○	○	●		○		○		○		○		○		○
Pesticides																
Total DDx (2,4 & 4,4)																
Dieldrin																
Total Chlordane		--		--		--		--		--		--		--		--
Hexachlorobenzene		--		--		--		--		--		--		--		--
Polycyclic Aromatic Hydrocarbons																
Total HMW PAH	--		--		--		--		--		--		--		--	
Total LMW PAH	--		--		--		--		--		--		--		--	
Total PAH		--		--		--		--		--		--		--		--
Inorganics																
Arsenic		--		--		--		--		--		--		--		--
Cadmium		--		--		--		--		--		--		--		--
Chromium		--		--		--		--		--		--		--		--
Copper																
Lead																○
Mercury										○		○		○		●
Methylmercury																
Nickel		--		--		--		--		--		--		--		--
Selenium	○	--	○	--	○	--	○	--	○	--	○	--	○	--	○	--
Silver		--		--		--		--		--		--		--		--
Zinc		--		--		--		--		--		--		--		--

Notes: ● = indicates LOAEL HQ ≥ 1; ○ = indicates NOAEL HQ ≥ 1; **-** = not evaluated in the LPR FFS; COPECs evaluated that did not have any exceedances are not shown.

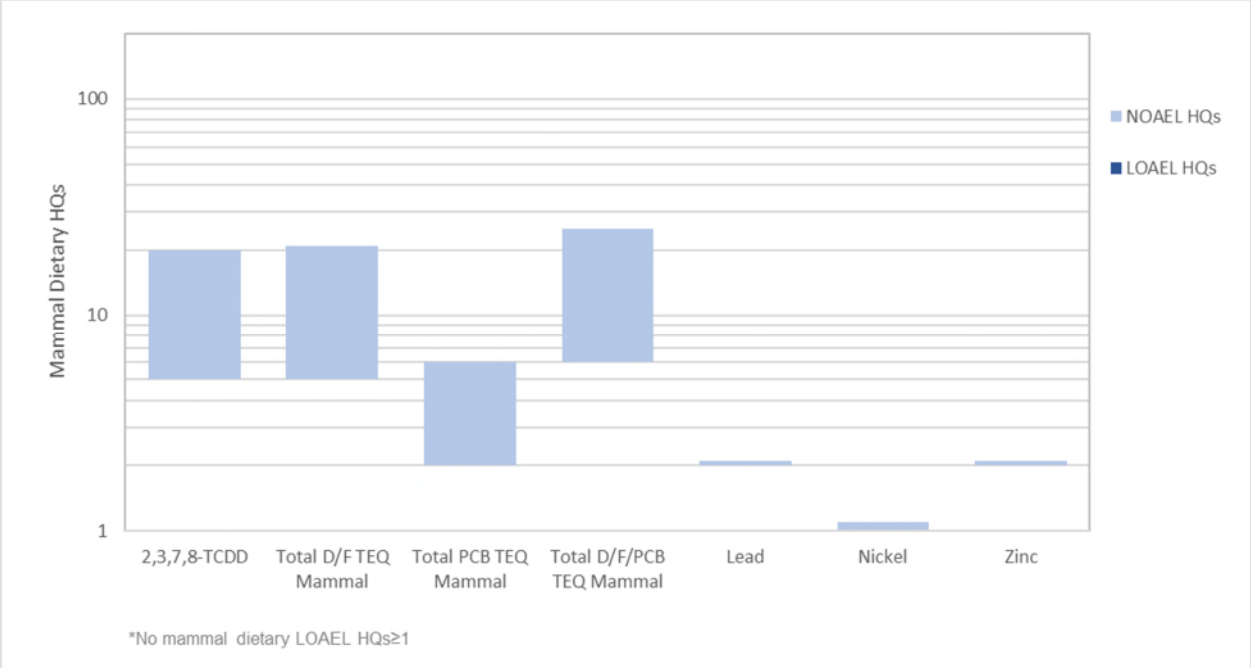
Table 9-5. Summary of TRV Exceedances for Mammal Dietary Assessment (cont.)

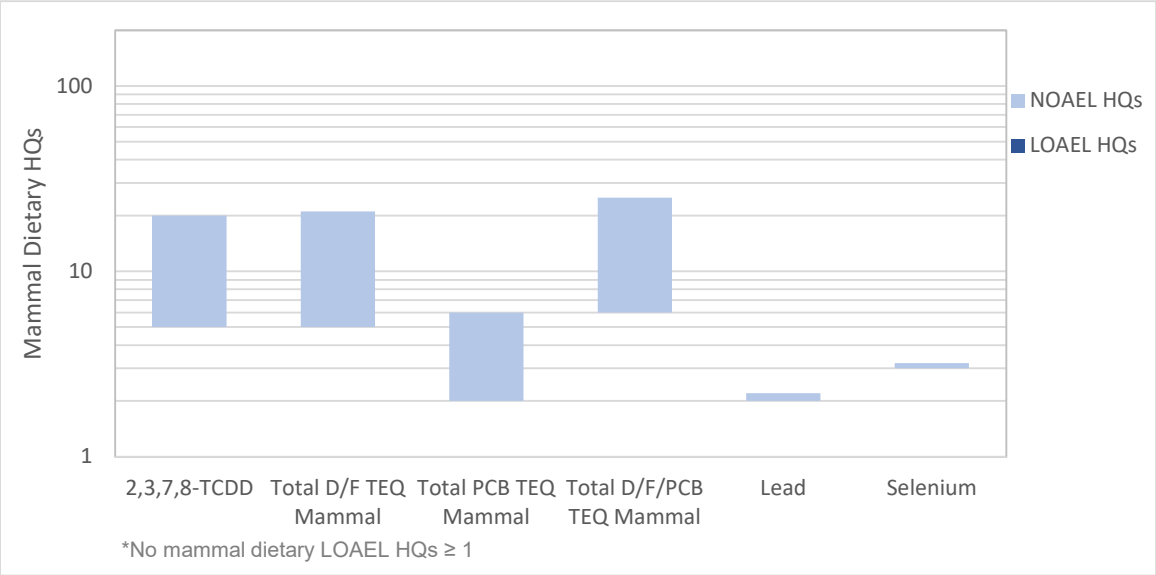
Preliminary COCs	NBSA and FFS LOAEL-based HQs ≥ 1.0							
	Harbor Seal							
	NBSA-wide		North		SE		SW	
	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS	NBSA	LPR FFS
Dioxins/Furans								
2,3,7,8-TCDD		○		○		○		○
Total Dioxin/Furan TEQ Mammal		○		○		○		○
Polychlorinated Biphenyls								
Total PCBs								
Total PCB TEQ Mammal		○		○		○		○
Total Dioxin/Furan/PCB TEQ Mammal		○		○		○		○
Pesticides								
Total DDx (2,4 & 4,4)								
Dieldrin								
Total Chlordane		--		--		--		--
Hexachlorobenzene		--		--		--		--
Polycyclic Aromatic Hydrocarbons								
Total HMW PAH	--		--		--		--	
Total LMW PAH	--		--		--		--	
Total PAH		--		--		--		--
Inorganics								
Arsenic		--		--		--		--
Cadmium		--		--		--		--
Chromium		--		--		--		--
Copper								
Lead								
Mercury								
Methylmercury								
Nickel		--		--		--		--
Selenium		--		--		--		--
Silver		--		--		--		--
Zinc		--		--		--		--

Notes: ● = indicates LOAEL HQ ≥ 1; ○ = indicates NOAEL HQ ≥ 1; **-** = not evaluated in the LPR FFS; COPECs evaluated that did not have any exceedances are not shown.

The range of HQs for the aggregate mammal dietary exposure estimates (i.e., all species) with HQs ≥ 1 are depicted in Figure 9-1~~. Muskrat and harbor~~ for the NBSA-wide exposure area. Harbor seal LOAEL HQs were less than 1 for all COPECs and exposure areas using NBSA- and LPR FFS-selected TRVs. Muskrat LOAEL HQs were less than 1 for all COPECS and exposure areas except for mercury in the Southwest Zone, where HQ=1 using the LPR FFS TRV. River otter LOAEL HQs are ≥ 1 in the North Zone using the LPR FFS TRVs ~~only~~for 2,3,7,8-TCDD, total dioxin/furan TEQ, and total dioxin/furan/PCB TEQ. River otter LOAEL HQs using NBSA-selected TRVs are all less than 1.

Figure 9-1. NBSA-wide Magnitude of NOAEL and LOAEL HQs ≥ 1 for Mammal Dietary Assessment





~~10.2~~9.2 Uncertainties Assessment

This section contains the uncertainties assessment for the mammal risk assessment. Both quantitative and qualitative characterizations are provided for the dietary assessment presented in Section 9.1. The objective of the uncertainty assessment is to provide perspective on the likely accuracy of the overall risk conclusions that are presented in Section 9.3.

Sources of uncertainty that influenced risk characterization for mammals included uncertainties in the analytical results, data evaluation, CSM, exposure assessment, effects assessment, and interpretation of the risk estimates, especially with respect to drawing conclusions regarding potential for unacceptable population-level risk. Many of these sources of uncertainty are generic in nature and inherent in the risk assessment process.

Dietary exposure modeling has been a well-established practice for wildlife (avian and mammalian) risk assessments (USEPA 1993; Sample et al. 1997) and oral doses are a common method of evaluating contaminant exposure. However, there are uncertainties specific to the dietary risk estimates for mammals, as discussed in this section. Inherent practices in risk assessment result in exposure estimates that are biased high in order to be reasonably conservative include:

- The use of an upper exposure value (i.e., 95% UCL) to calculate dietary exposure concentrations
- The use of a maximum values in the place of a UCL when limited data are available
- The assumption that a mammal species feeds exclusively or is getting all of its seasonal and lifetime exposure to COPECs, from the NBSA (i.e., SUF = 1)
- The assumption that COPECs are 100% bioaccessible via dietary uptake (dermal absorption via the gut).

As a result, risk estimates are expected to be overestimated rather than underestimated in this BERA. For risk estimates based on dietary HQs, TRVs are a key source of uncertainty in the risk estimates and are discussed below or in Appendix D, as relevant. Additional sources of uncertainty were identified and evaluated either quantitatively or qualitatively. Table 9-6 summarizes the quantitative uncertainty evaluations conducted for mammals for the dietary assessments and their associated effect on risk estimates. Risk calculations and HQs based on the quantitative uncertainty evaluations are presented in Appendix F and discussed in the following sections. Qualitative evaluations were also conducted and are presented below in their respective sections.

Table 9-6. Summary of Quantitative Mammal Uncertainty Evaluations

Species	General Uncertainty Evaluated	Specific Evaluation Conducted	Difference in HQ Values	Effect on Risk Conclusions[a]
Mammal Dietary Assessment				
River otter~~,~~ ~~Harbor seal~~	SUF of 0.5	HQs recalculated using SUF of 1	HQs increase (100%)	~~River otter[b]:~~NBSA-wide: Increase HQ to ≥ 1 for ~~Total PCB based on NBSA TRVs; Total PCBs,~~ 2,3,7,8-TCDD, Total ~~TEQ – Mammal, and Total PCB/~~Dioxin~~-~~/Furan TEQ, Total PCBs, Total Dioxin/Furan/PCB TEQ ~~– Mammal~~ based on LPR FFS TRVs and for Total PCBs based on NBSA TRVs. ~~Harbor seal: no change~~N zone: Increase HQ to ≥ 1 for based on NBSA and LPR FFS TRVs for Total PCBs.

				SE zone: Increase HQ to ≥ 1 for 2,3,7,8-TCDD, Total PCBs, Total Dioxin/Furan/PCB TEQ based on LPR FFS TRVs and for Total PCBs based on NBSA TRVs. SW zone: Increase HQ to ≥ 1 for Total PCBs, Total Dioxin/Furan/PCB TEQ based on LPR FFS TRVs and for Total PCBs based on NBSA TRVs.
Harbor seal	SUF of 0.5	HQs recalculated using SUF of 1	HQs increase (50%)	No change
River otter	NBSA-wide Polychaete worm EPCs used	Representative COPEC re-evaluated using sediment-to-worm regression and zone sediment EPCs	HQs decrease (<5%)	No change
All	Treatment of NDs as zero for EPC calculation	EPCs recalculated using half or full ~~DL~~QL	~~HQs[b,c]~~HQs increase (<0.1)	River otter NBSA-wide increase HQ to ≥ 1 when using LPR FFS TRVs and ND=1/~~2~~2QL or ~~1~~1QL for Total Dioxin/Furan TEQ ~~– mammal~~ and Total Dioxin/Furan/PCB~~/Dioxin~~ TEQ ~~– Mammal~~
River otter; Harbor seal	Fish prey size of <30 cm assumed	EPCs recalculated using all prey fish	~~HQs[b]~~HQs mostly decrease (13-92%), some increase (10-100%)	No change

~~**Notes**:~~
~~a. Based on LOAEL HQs ≥ 1.~~
~~b. NBSA-wide only results presented in this table. Differences for assessment zones are discussed in the text. All results of the sensitivity analyses are provided in Appendix F.~~
~~c. For COPECs with LOAEL HQs ≥ 1 only.~~
Notes: ND – non detected; QL – quantitation limit

Additionally, relatively large exposure areas were evaluated in this BERA, including the NBSA-wide area and individual assessment zones. While these areas are appropriate for risk assessment, especially for population-level assessment of mammal communities, evaluation of smaller features, such as individual wetlands or mudflats, may be required in the FS. The uncertainty associated with applying the results of the NBSA-wide and assessment zone conclusions to localized areas within the NBSA is considered to be low for highly mobile receptors, such as most mammals, and somewhat higher for species restricted to specific habitat types, such as muskrat. For muskrat, this uncertainty was reduced by evaluating shoreline areas instead of the larger assessment zones.

~~10.2.1~~9.2.1 Mammalian TRVs

A key uncertainty associated with the mammalian dietary risk characterization is the selection of TRVs used in the risk calculations. As noted in Section 3.5 and at the request of USEPA, two sets of TRVs were

evaluated to maintain consistency and allow comparison between the LPRSA, LPR FFS, and NBSA BERAs.

In general, TRVs are selected to be protective of either the 5th percentile of an SSD or the lowest acceptable LOAEL based on single studies reviewed. Thus, they tend to err on the side of over predicting risk. The TRVs for dioxins/furans (2,3,7,8-TCDD and TEQ), were further evaluated as a key uncertainty in the mammal assessment because they were the ~~only~~ COPECs for which LOAEL HQs ≥ 1 were most frequently predicted. General uncertainties associated with TRVs are discussed in Appendix D.

The NBSA TRVs for 2,3,7,8-TCDD and TEQs are based on reproductive effects in mink (Hochstein et al. 2001) and are the same as was used in the LPRSA BERA (Windward 2019). The LPR FFS TRVs are also based on reproductive effects in mink (Tillitt et al. 1996); however, the LPR FFS LOAEL (2.2 ng/kg-day) is approximately the same as the NBSA NOAEL (2.6 ng/kg-day). The two studies identified very different toxicity curves with a factor of 28 separating the NOAEL and LOAEL in the study used as the basis for the LPR FFS TRV (Tillitt et al. 1996), and a factor of 3 separating the NOAEL and LOAEL in the study used as the TRV basis in the LPRSA BERA (Hochstein et al. 2001). Beyond this single species, the sensitivity of mammals in general to TCDD toxicity is highly variable with acute toxicity ranging among species up to a factor of 8,400 between the single oral lethal dose, 50% (LD50) dose for the guinea pig (the most sensitive mammal) and the hamster (Eisler 1985).

The primary reason why TRVs were developed in the LPRSA BERA for mink based on the Hochstein et al. (2001) study was that this was a laboratory study with 2,3,7,8-TCDD incorporated into the diet while the Tillitt et al. (1996) study fed mink field contaminated fish with variable levels of multiple COPECs in the diet including PCBs, dioxins/furans, and multiple other contaminants. The use of a food source containing multiple contaminants ~~makes the LPR FFS TRV more uncertain than the NBSA TRV~~introduces significant uncertainty for evaluating effects associated with only 2,3,7,8-TCDD, as the effects seen in the mink in the Tillitt et al. (1996) study could have been caused by synergistic effects of contaminant mixtures ~~(Table 9-7).~~. The toxicity study by Tillitt et al. (1996) is preferred for evaluating ecological risks specific to the region where the study was conducted because it captures unique aspects of the ecosystem that can affect toxicity: multiple contaminants, heterogeneous genetic composition, contaminant bioavailability, and non-contaminant potential stressors such as disease, weather, and food availability. For the NBSA area, site-specific field toxicity studies are not available, and the extent to which the contaminant mixture evaluated by Tillitt et al. (1996) is representative of the contaminant mixture in the NBSA area is uncertain. Exposure to dioxins, furans, dioxin-like PCBs in this BERA was evaluated using the 2,3,7,8-TCDD TEQ approach, in which concentrations of individual constituents are adjusted based on relative toxicity to a 2,3,7,8-TCDD. Therefore, use of the TRVs based on only 2,3,7,8-TCDD (Hochstein et al. 2001) is ~~more~~considered appropriate for use with the TEQ approach used in this BERA~~, and the Hochstein et al. (2001) TRVs are considered to produce risk estimates with lower uncertainty than the Tillitt et al. (1996) TRVs~~. While site-specific field studies such as Tillitt et al. (1996) are preferred over laboratory studies for evaluating toxicity of a particular site, and can provide information useful for bounding risk estimates, they are not generally preferred over laboratory studies at different sites because of the great inherent number of variables that can influence the toxicity between sites.

~~Table 9-7. Assessment of Mammalian TRV Reliability for Risk Estimation~~

~~Preliminary COCs~~	~~Reliability for Risk Estimation in the NBSA BERA~~

	~~Dietary TRVs~~	~~Dietary TRVs~~
	~~NBSA TRVs~~	~~LPR FFS TRVs~~
~~Dioxins/Furans & TEQ[a]~~	~~Moderate; Lowest LOAEL[c]~~	~~Low; Lowest LOAEL[b]~~

~~**Notes:**~~
~~a. TEQ related COPECs include 2,3,7,8-TCDD, Total Dioxin TEQ Mammal, and Total Dioxin/Furan/PCB TEQ Mammal.~~
~~b. Fish diet contained co-contaminants.~~
~~c. This value was also selected in the LPRSA BERA (Windward 2019).~~

Regarding the applicability of TRVs from a mink study to be applicable to the mammalian receptors in this assessment (river otter, muskrat, and harbor seal), based on taxonomic relatedness, the sensitivity of river otters is expected to be similar since river otter and mink are both in the family Mustelidae. Muskrats are in the same taxonomic order (Rodentia) as guinea pigs (the most sensitive species found to 2,3,7,8-TCDD with a LOAEL of 4.9 ng/kg-day as compared to the mink LOAEL of 8.8 ng/kg-day as reported in the LPRSA BERA) and rats (which were found to be less sensitive than mink to 2,3,7,8-TCDD as reported in the LPRSA BERA). The degree to which a mink TRV is protective of harbor seals is not known.

~~10.2.2~~9.2.2 Exposure Assumptions

In this section, uncertainties specific to the NBSA dietary exposure assessment were identified and were evaluated quantitatively or qualitatively to assess potential impact on the risk estimates presented above.

Uncertainties related to the selected SUFs, polychaete worm EPCs, treatment of non-detects in calculated mixture totals (e.g., total PCBs), and prey fish size were quantitatively evaluated. Other uncertainties, discussed below, were qualitatively assessed.

~~10.2.2.1~~9.2.2.1 Site Use Factors

The selection of SUFs for mammals and the related question of whether there is habitat for mammals are important factors in understanding risk estimates for mammals in the NBSA. A SUF of 0.5 was applied to exposure estimates for the river otter and harbor seal. The SUF was selected based on data from the literature indicating that these species have large home ranges compared to the size of the NBSA and prefer habitats for foraging (tidal mud flats, sand bars, shoals, river deltas, estuaries, bays, coastal rocks, and offshore islets; USEPA 1993) and loafing (cobble and sand beaches, tidal mud flats, offshore rocks and reefs, glacial and sea ice, and man-made objects such as piers and log booms; USEPA 1993) that are not found in the NBSA or make up a very small proportion of the NBSA. Use of the SUF is reasonable and appropriate based on the habitat preferences of these species. The use of an alternate SUF of 1 was examined for the uncertainty assessment and results in additional COPECs with LOAEL HQs ≥ 1. These include:

- For river otter, when dietary food web models assumed a SUF =1 rather than 0.5, the following changes to HQs occurred:
  - NBSA-wide HQs:
    - NBSA TRVs: total PCB LOAEL HQ changed from < 1 to ≥ 1.

    - LPR FFS TRVs: total PCBs, 2,3,7,8-TCDD, total dioxin/furan TEQ – mammal, and total ~~PCB~~ dioxin/furan/PCB TEQ – mammal LOAEL HQs changed from < 1 to ≥ 1.
  - North Zone HQs:
    - NBSA TRVs: total PCBs LOAEL HQs changed from < 1 to ≥ 1.
    - LPR FFS TRVs: The total PCBs LOAEL HQs changed from < 1 to ≥ 1.
  - Southeast Zone HQs:
    - LPR FFS TRVs: 2,3,7,8-TCDD, total dioxin/furan TEQ – mammal, and total ~~PCB~~ dioxin/furan/PCB TEQ – mammal LOAEL HQs changed from < 1 to ≥ 1.
  - Southwest Zone HQs:
    - NBSA TRVs: total PCBs LOAEL HQs changed from < 1 to ≥ 1
    - LPR FFS TRVs: total PCBs and total dioxin/furan/PCB ~~Dioxin~~ TEQ - mammal LOAEL HQs changed from < 1 to ≥ 1.
- Harbor seal food web models also assumed that SUF = 0.5, given the lack of haul-out sites for this receptor in the NBSA. When harbor seal food web models were analyzed with SUF = 1, there were no changes from HQ < 1 to HQ ≥ 1 for NBSA-wide or zone LOAEL HQs based on NBSA or LPR FFS TRVs.

As noted above, the SUFs evaluated in this BERA are appropriate based on the available information for the NBSA. HQs estimated using SUF of 1 for river otter and harbor seal are provided only to demonstrate the effect of application of an SUF on the HQs; use of a SUF of 1 is not warranted in this BERA. Summary tables of HQ results from this sensitivity analysis on the effect of SUF on mammal dietary risk are found in Appendix F.

~~10.2.2.2~~9.2.2.2 Polychaete Worm EPCs

The polychaete worm tissue data used in the food web models were based on the laboratory bioaccumulation study in which worms were exposed to homogenized sediment collected from the 0 to 15-cm depth horizon. Uncertainties related the representativeness of these bioaccumulation tests for estimating dietary exposures are discussed in Section 7.6.3.3. River otter included polychaete worm as a portion of its diet; however, polychaete worm tissue sample size was not large enough to be evaluated by assessment zone. Therefore, risk from ingestion of polychaete worms were assessed NBSA-wide in the zone risk calculations for this species. A regression analysis was performed to assess the relationship between sediment concentrations and polychaete worm tissue concentrations, using 2,3,7,8-TCDD as an example COPEC. There was a strongly significant relationship between sediment and polychaete worm tissue 2,3,7,8-TCDD concentrations ($r^2 = 0.91$, $p=0.0002$; Figure 8-2). The regression equation from this analysis was used to predict polychaete worm 2,3,7,8-TCDD concentration from sediment concentrations. The predicted polychaete worm tissue concentrations were then used in the food web models used to calculate NBSA-wide and assessment zone risk HQs. There was no change in 2,3,7,8-TCDD HQs for river otter when using the actual polychaete worm tissue data or the predicted worm tissue data derived from the relationship between sediment and worm tissue concentrations. This is not unexpected given the relatively small dietary exposure attributable to invertebrates in the mink diet. The regression analysis

assumed that the sediments used in the laboratory bioaccumulation study are representative of conditions throughout the NBSA. Summary tables of HQ results from this sensitivity analysis on the effect of using sediment to worm tissue regressions on risk from dietary exposure in mammals are presented in Appendix F.

~~10.2.2.3~~9.2.2.3 Treatment of Non-Detects

The concentrations of sum components (e.g., PCB congeners) that were not detected were assumed to be zero when calculating totals. The effect on HQs of using one-half the ~~detection~~quantitation limit or the full ~~detection~~quantitation limit was evaluated.

NBSA-wide LOAEL HQs based on LPR FFS TRVs changed from < 1 to ≥ 1 when using ND = ½ and 1 for ~~Total~~total dioxin/furan TEQ – Mammal and ~~Total~~ total dioxin/furan/PCB~~/Dioxin~~ TEQ - Mammal for river otter.

There were no other changes from HQ < 1 to HQ ≥ 1 for NBSA-wide and zone HQs based on NBSA or LPR FFS TRVs regardless of how non-detected values in sums were treated (either as zero, one-half the ~~detection~~quantitation limit, or the full ~~detection~~quantitation limit). Summary tables of HQ results from this sensitivity analysis on the treatment of non-detects are presented in Appendix F.

~~10.2.2.4~~9.2.2.4 Fish Prey Size

Based on data from the literature, only fish < 30 cm were included in the fish dietary components for river otter (80% of diet included fish < 30 cm) and harbor seal (90% of diet included fish < 30 cm). This diet excluded several larger fish samples.

(i) Fish Prey Size – fish < 30 cm and all fish

River otter and harbor seal dietary food web models were analyzed using all fish samples to assess this uncertainty. There were no changes from LOAEL HQ < 1 to HQ ≥ 1 for NBSA and LPR FFS TRVs, either NBSA-wide or by zone, for either species. The calculated EPCs for fish of all lengths used and the summary tables of HQ results from this sensitivity analysis on the effect of including fish of all lengths in the mammal dietary risk evaluation are presented in Appendix F.

(i) Fish Prey Size – fish < 30 cm and fish > 30 cm

River otter dietary food web models were analyzed using fish > 30 cm to assess this uncertainty. LOAEL HQs based on LPR FFS TRVs for 2,3,7,8-TCDD, total dioxin/furan TEQ - mammal, and total dioxin/furan/PCB TEQ – Mammal decreased from ≥ 1 to < 1 in the North Zone when 100% of the otter diet was assumed to include only fish > 30 cm. No changes in HQs were found when the river otter diet was 5% polychaete worm, 10% softshell clam, 5% blue crab, 70% fish < 30cm, and 10% fish > 30cm or when the river otter diet was 5% polychaete worm, 10% softshell clam, 5% blue crab, 75% fish < 30cm, and 5% fish > 30cm.

The calculated EPCs for fish > 30 cm in length used are presented in Appendix C and the summary tables of HQ results from this sensitivity analysis on the effect of including fish > 30 cm in the river otter dietary risk evaluation are presented in Appendix F.

~~10.2.2.5~~9.2.2.5Other Uncertainties

Additional uncertainties were identified that are not expected to substantially impact the risk estimates. These uncertainties were identified but not quantitatively evaluated:

- Modeled diet – Dietary items were limited to prey species with tissue chemistry data available from the NBSA, including worm, blue crab, softshell clam, and fish tissue, and did not include other prey items that may be additional components of mammal diets, such as zooplankton, squid, shrimp, mammals, amphibians, or aquatic and terrestrial plants. In addition to the limited types of prey used in the mammal dietary model, the selection of explicit prey portions may not reflect the feeding behavior of some mammal species. Therefore, the representativeness of the dietary estimates (based on available prey tissue data) for actual NBSA mammal diets is uncertain. However, as prey tissue concentrations were measured in species with relatively high bioaccumulation potential (e.g., infaunal life history, higher trophic level consumers) compared with many of the unmeasured dietary components, it is likely that COPEC exposure via the diet is overestimated rather than underestimated in this BERA.
- Use of soil to plant BAFs to estimate concentrations of COPECs in aquatic plant tissue for muskrat – Aquatic plants make up a significant portion (85%) of the muskrat diet; however, aquatic plant tissue data are not available from the NBSA. Additionally, sediment to aquatic plant BAFs are generally unavailable. Therefore, soil to plant BAFs from the literature were used to estimate the concentration of COCs in plant tissue. For many of the key COPECs in the NBSA (PCBs, dioxins/furans, many metals), plant uptake from soil is lower than for invertebrates and/or vertebrates exposed to the same media. However, it is unknown whether the BAFs accurately reflect the uptake of COPECs in aquatic plants or whether the resulting values represent actual COPEC aquatic plant tissue concentrations available to muskrats in the NBSA.
- Food ingestion rate – Food ingestion rates for river otter and muskrat were estimated as a function of body weight using equations from Nagy (2001). The food ingestion rate for harbor seal was based on a model developed by Ashwell-Erickson and Elsner (1981) that was also a function of body weight (cited in USEPA 1993). It is unknown how closely the modeled food ingestion rates reflect actual ingestion rates of NBSA mammals, but these studies have derived these rates from empirical studies on these mammals. The use of these ingestion rate equations for mammals is standard practice in Superfund risk assessments, and it is not likely that ingestion rates vary substantially in a manner that would affect risk outcomes. The CPG conducted a probabilistic sensitivity analysis of various exposure factors for mink and otter in the LPRSA BERA. In general, they found that the deterministic risk calculations were generally more conservative, and produced higher HQs. These results suggest an added level of conservatism and certainty in the NBSA risk estimates from the deterministic food web model.
- Sediment ingestion rate – Measured incidental sediment ingestion rates were not available for any of the mammal receptors. Muskrat was assumed to have the same sediment ingestion rate as hispid cotton rat (Garten 1980). Sediment ingestion rates for the other species were based on what is known about each species' feeding habitats and best professional judgment. It is unknown whether the estimated sediment ingestion rates reflect actual ingestion rates of NBSA mammals. For bioaccumulative COPECs, prey uptake is assumed to account for the majority of dietary exposure,

and therefore uncertainty in the sediment ingestion rate is unlikely to influence risk conclusions for mammals in this BERA.

~~10.3~~9.3 Weight of Evidence Risk Findings for Mammals

As summarized in Section 3.5, a WOE approach was used to evaluate COPECs with LOAEL HQs ≥ 1 (preliminary COCs; Table 9-7) and potential for unacceptable risk to mammal populations in the NBSA.

Table 9-7. Summary of Preliminary COCs for Mammal Dietary Assessment

Preliminary COCs	Exposure Area	Range of LOAEL HQs					
		Mammal Diet LOE					
		NBSA TRVs			LPR FFS TRVs		
		River Otter	Muskrat	Harbor Seal	River Otter	Muskrat	Harbor Seal
2,3,7,8-TCDD	NBSA-wide	<1	<1	<1	<1	<1	<1
	Zones	<1	<1	<1	<1- **1**	<1	<1
Total Dioxin/Furan TEQ Mammal	NBSA-wide	<1	<1	<1	<1	<1	<1
	Zones	<1	<1	<1	<1 - **1**	<1	<1
Total Dioxin/Furan/PCB TEQ Mammal	NBSA-wide	<1	<1	<1	<1	<1	<1
	Zones	<1	<1	<1	<1 - **1**	<1	<1
Mercury	NBSA-wide	<1	<1	<1	<1	<1	<1
	Zones	<1	<1	<1	<1	<1 - **1**	<1

For mammals, a range of dietary HQs (including HQs based on sensitivity analyses) provides the LOE used in this BERA. The range of HQs includes:

- HQs based on dietary doses in wildlife food web models estimated using measured prey tissue concentrations compared to literature-derived dietary TRVs specific to the NBSA
- HQs based on dietary doses in wildlife food web models estimated using measured prey tissue concentrations compared to literature-derived dietary TRVs derived for the LPR FFS (USEPA 2014).

HQs were calculated for the NBSA (sitewide for river otter and harbor seal; shorelines habitat for muskrat), and also independently for the north, southeast, and southwest assessment zones. A variety of sensitivity analyses for the dietary HQs were conducted as well. Secondary LOEs, such as reference comparisons or site-specific toxicity data, were not available for mammals.

Uncertainties in the HQs, discussed in Section 9.2, were evaluated in the WOE assessment according to the factors presented in Section 3.5 to arrive at an overall risk conclusion for each preliminary COC. The risk conclusions based on the WOE are presented in Table 9-8 and discussed below.

As discussed above in Section 9.2, mammalian TRVs are a key uncertainty in the risk estimates and apply to all mammalian receptors evaluated in this BERA. However, the two sets of TRVs were included for bounding purposes and given equal weight in this WOE evaluation. Additional LOE are discussed below for each receptor and preliminary COC.

~~10.3.1~~9.3.1 River Otter

For river otter, ~~no~~ dietary LOAEL HQs ≥ 1 were predicted for ~~any~~ COPECs ~~based on~~ presented in Table 9-9. Sensitivity analyses conducted for this receptor included evaluation of ~~the NBSA-wide area, or~~ NDs for the dietary HQs, modified prey size assumptions for the ~~southeast or southwest assessment zones using either set~~dietary HQs, use of ~~TRVs.~~an uptake model for polychaete worm dietary fraction, and evaluation of an SUF of 1.

Using the primary exposure and toxicity assumptions, dietary LOAEL HQs ≥ 1 were predicted only for 2,3,7,8-TCDD, total dioxin/furan TEQ, and total dioxin/furan/PCB TEQ in the North Zone using the LPR FFS TRVs. Sensitivity analyses, including evaluation of a SUF of 1 and calculation of total concentrations using the half and full quantitation limit for NDs resulted in identification of additional assessment zones for some of the preliminary COCs and total PCBs as an additional preliminary COC based on LOAEL HQs equal to 1 for the NBSA-wide area and/or one or more individual assessment zones.

Table 9-9. Exceedances of TRVs for River Otter Dietary Assessment

Preliminary COCs	Dietary HQs ≥ 1	
	NBSA TRVs	LPR FFS TRVs
Primary Exposure and Toxicity Assumptions (Table 9-5)		
2,3,7,8-TCDD	None	N zone
Total Dioxin/Furan TEQ	None	N zone
Total Dioxin/Furan/PCB TEQ	None	N zone
Sensitivity Analyses (Table 9-6)		
2,3,7,8-TCDD	None	NBSA-wide, N, SE zone (SUF=1)
Total Dioxin/Furan TEQ	None	NBSA-wide, N, SE zone (SUF=1); NBSA-wide (NDs = ½QL)
Total Dioxin/Furan/PCB TEQ	None	NBSA-wide, N, SE, SW zones (SUF=1); NSBA-wide (ND=1/2QL, ND=QL).
Total PCBs	NBSA-wide, N, SW zones (SUF=1)	NBSA-wide, N, SW zones (SUF=1)

Notes: ND – non detected; QL – quantitation limit

Inputs to the HQs, including SUFs, EPCs, and TRVs, are discussed below as part of the WOE evaluation for this receptor.

SUF: For river otter, the NBSA-wide area represents the most realistic exposure scenario, as the individual assessment zones (1.4, 1.0, and 2.1 square miles for the north, southeast, and southwest assessment zones, respectively) are small relative to the home range of an individual river otter (range 3-30 square miles; Helon et al. 2004; Wilson 2012). Even the NBSA-wide area (6.3 square miles) is unlikely to be large enough to accommodate a population of river otters, and no river otters have been observed in the NBSA, either as noted in historic studies or during the 2013 reconnaissance survey (Tierra 2015b). As such, it is reasonable to assume that the SUF of 0.5 represents a conservative estimate of time spent

in the NBSA (both NBSA-wide and specifically for the north assessment zone). A lower SUF is likely appropriate.

Diet: The prey diet evaluated in this BERA consists primarily of fish, with some of invertebrates with high potential for uptake. The diet evaluated is considered unlikely to significantly over- or underestimate exposure.

EPCs: In general, EPCs were based on 95%UCL concentrations which are appropriate exposure estimates for a BERA.

Based on the various factors considered in the dietary HQ LOE, the resulting risk conclusions for preliminary COCs are as follows:

- Dioxins/furans and TEQ: Unacceptable risk is unlikely. This conclusion is based on 1) LOAEL HQs < 1 using NBSA TRVs; 2) low magnitude of the LOAEL HQs ≥ 1 using LPR FFS TRVs; 3) conservative assumptions used in the risk estimates (e.g., conservative TRVs and SUFs; and 4) LOAEL HQs < 1 using both sets of TRVs and most realistic assumptions for river otter populations (primary exposure assumptions and NBSA-wide area).
- Total PCBs: Unacceptable risk is unlikely. This conclusion is based on 1) LOAEL HQs < 1 using both sets of TRVs and most realistic assumptions for river otter populations (primary exposure assumptions and NBSA-wide area); 2) low magnitude of the LOAEL HQs ≥ 1 using SUF of 1; 3) conservative assumptions used in the risk estimates (e.g., conservative TRVs and SUFs)

~~10.3.2~~9.3.2 Muskrat

~~No~~For muskrat, dietary LOAEL HQs ≥ 1 were predicted for COPECs presented in Table 9-10. Sensitivity analyses conducted for this receptor included evaluation of NDs for the dietary assessment.

Using the primary exposure and toxicity assumptions, dietary LOAEL HQs ≥ 1 were predicted only for mercury in the southwest zone based on LPR FFS TRVs. No other LOAEL HQs were ≥ 1 for the NBSA-wide, North and Southeast Zones were predicted for muskrat potentially inhabiting shoreline areas ~~within the NBSA~~based on the primary exposure assumptions or ~~within any of the~~ sensitivity analyses.

Table 9-10. Exceedances of TRVs for Muskrat Dietary Assessment

Preliminary COCs	Dietary HQs ≥ 1	
	NBSA TRVs	LPR FFS TRVs
Primary Exposure and Toxicity Assumptions (Table 9-5)		
Mercury	None	SW zone
Sensitivity Analyses (Table 9-6)		
No additional COCs were identified from sensitivity analyses for this receptor		

Inputs to the HQs, including SUFs, EPCs, and TRVs, are discussed below as part of the WOE evaluation for this receptor.

SUF: For muskrat, a SUF of 1 is appropriate based on the small home range for this species. Although habitat is found predominantly along the shoreline, the NBSA-wide area and individual assessment

zones~~. Uncertainties related to the risk~~ exceed the home range (1 to 12 acres) for an individual muskrat (Miller 2018). The shoreline habitat within the NBSA-wide area (6.3 square miles) is unlikely to be large enough to accommodate a population of muskrats; therefore, the NBSA-wide exposure area is considered more relevant than the smaller individual zones.

Diet: The diet evaluated in this BERA consists primarily of aquatic plant material, with a small proportion of invertebrates and fish with higher potential for uptake. The diet evaluated is considered unlikely to significantly over- or underestimate exposure.

EPCs: In general, EPCs were based on 95%UCL concentrations which are appropriate exposure estimates for ~~muskrat are not likely to result in changes to the~~a BERA.

Based on the various factors considered in the dietary HQ LOE, the resulting risk conclusions for ~~this receptor. No unacceptable~~the preliminary COC are as follows:

- Mercury: Unacceptable risk is unlikely. This conclusion is based on 1) LOAEL HQs < 1 for most scenarios; 2) low magnitude of the LOAEL HQs ≥ 1; 3) conservative assumptions used in the risk ~~was predicted for muskrat, and no COCs are identified for this receptor.~~estimates (e.g., conservative TRVs and SUFs; and 4) LOAEL HQs < 1 using both sets of TRVs and most realistic assumptions for muskrat populations (primary exposure assumptions and NBSA-wide area).

~~10.3.3~~9.3.3 Harbor Seal

No LOAEL HQs ≥ 1 were predicted for harbor seal potentially inhabiting the NBSA or any of the assessment zones~~.~~ (Table 9-11). Sensitivity analyses conducted for this receptor included evaluation of NDs, dietary HQs, modified prey size assumptions for the dietary HQs, and evaluation of an SUF of 1.

Table 9-11. Exceedances of TRVs for Harbor Seal Dietary Assessment

Preliminary COCs	Dietary HQs ≥ 1	
	NBSA TRVs	LPR FFS TRVs
Primary Exposure and Toxicity Assumptions (Table 9-5)		
No COCs were identified for this receptor		
Sensitivity Analyses (Table 9-6)		
No additional COCs were identified from sensitivity analyses for this receptor		

Uncertainties related to the risk estimates for harbor seal are not likely to result in changes to the risk conclusions for this receptor. No unacceptable risk was predicted for harbor seal, and no COCs are identified for this receptor.

~~10.3.4~~9.3.4 Summary of Risk Findings for Mammals

Overall, no unacceptable risk to populations of mammals potentially exposed to COPECs in the NBSA is expected. Conclusions for the mammal evaluations are as follows:

- ~~For muskrat (omnivorous/herbivorous mammal) and~~For harbor seal (piscivorous aquatic mammal): LOAEL HQs < 1 are predicted for all scenarios evaluated in the NBSA-wide and assessment zone areas.
- ~~For river otter:~~
- For muskrat (omnivorous/herbivorous mammal):
  - LOAEL HQs < 1 are predicted for all scenarios evaluated in this BERA except for mercury in the southwest zone using LPR FFS TRVs.
- For river otter:
  - Using primary exposure and toxicity assumptions, LOAEL HQs < 1 are predicted for all scenarios evaluated in this BERA except for dioxins/furans and TEQ using LPR FFS TRVs.
  - Sensitivity analyses, including evaluation of a SUF of 1 and calculation of total concentrations using the full quantitation limit for NDs, resulted in identification of total PCBs as an additional preliminary COC based on LOAEL HQs equal to 1 for the NBSA-wide area and/or one or more individual assessment zones.
  - Unacceptable risk to these COPECs is unlikely based on LOAEL HQs < 1 using most realistic assumptions for river otter populations (NBSA TRVs and both sets of TRVs in NBSA-wide area); conservative assumptions used in the risk estimates (e.g., conservative TRVs and SUFs; and low magnitude of the LOAEL HQs ≥ 1.~~ LOAEL HQs ≥ 1 were predicted only for dioxins/furans (2,3,7,8-TCDD, total mammalian TEQ, and total PCB dioxin mammal TEQ) in the north assessment zone using LPR FFS TRVs.~~

Based on the results of this BERA, potential exposures to COPECs in the NBSA are not expected to pose unacceptable risk to mammals. ~~The possible, but unlikely exception, is limited potential risk to river otter from 2,3,7,8-TCDD and total TEQs if they were utilizing the North Zone.~~ As summarized in Table 9-8, risk drivers/COCs (identified based on LOAEL HQs ≥1 and WOE supporting the conclusion of unacceptable risk) were not identified for mammals.

~~11~~10 DIAMONDBACK TERRAPIN (REPTILE) ASSESSMENT

Reptiles, including the diamondback terrapin (*Malaclemys terrapin*), were evaluated qualitatively in this BERA, because of their limited potential occurrence and the lack of available species-specific TRVs. Available field survey data are insufficient to determine the presence or absence of the northern diamondback terrapin in Newark Bay. USACE (1997) notes the possibility that the diamondback terrapin could occur as a transient visitor to the NBSA as it moves between the Hackensack Meadowlands Complex and marshes and tidal creeks along the Arthur Kill during the spring and summer. The diamondback terrapin is the only species of turtle in North America that spends its life in brackish water (National Aquarium 2010). No terrapins have been collected to date in bottom trawls, but they appear to be a pollution-tolerant species (Wood 1995, as cited in USACE 1997). USACE (1997) also notes the possibility for three species of marine turtles – loggerhead (*Caretta caretta*), Kemp's ridley (*Lepidochelys kempii*), and the green sea turtle (*Chelonias mydas*) – to utilize the shoal areas (i.e., subtidal flats) of Newark Bay. However, there is no documented evidence of these species in the NBSA.

~~11.1~~10.1 Qualitative Risk Characterization

The exposure of terrapins or other reptiles in the NBSA would likely be comparable to that for the mammals assessed in Section 9. The terrapin is dependent upon saltmarsh and estuarine habitats in the temperate zone and requires sandy upland substrate free from predation for egg laying (Hart and Lee 2006). Available habitat for these receptors in terms of intertidal wetland areas in the Bay are small, representing only 0.7% of the total area of the NBSA. As such, it is likely that they would be exposed only on a limited basis.

Exposures for reptiles would be from surface water, sediments, and the food web. Risks to these receptors are expected to be *de minimis* based on the following:

- The surface water assessment found no unacceptable concentrations of COPECs in the NBSA either spatially, temporally, or seasonally
- No risks (i.e., LOAEL HQs >=1) were found for any of the mammal species assessed in this BERA
- The CPG (Windward 2019) undertook a quantitative assessment of risks to reptiles for the LPRSA BERA (Appendix N of the LPRSA BERA) for COPECs that had sufficient toxicological data (chromium, copper, lead, mercury, nickel, silver, and zinc) and found that all HQs were less than 1.

For these reasons, it is assumed there are no risks to reptiles in the NBSA.

~~11.2~~10.2 Uncertainties Assessment

The potential risks to all reptiles in the NBSA are uncertain due to lack of toxicological reference data. The CPG (Windward 2019) undertook a quantitative assessment of risks to reptiles for the LPRSA BERA for chromium, copper, lead, mercury, nickel, silver, and zinc and found that all HQs were less than 1. Potential risks from other COPECs could not be determined because toxicological data are not sufficient. No relevant toxicological data have been published since the LPRSA BERA to alleviate this uncertainty.

The degree of uncertainty in the qualitative reptile assessment is assumed to be low. This is due to the fact that: 1) habitat-based exposures in the NBSA are likely limited for these organisms; 2) surface water exposure to COPECs is negligible; and, 3) the results of the mammals risk assessment (which are comparable in terms of exposure-risk potential) do not demonstrate risk.

~~12~~11 RISK SUMMARY AND CONCLUSIONS

This BERA evaluated five assessment endpoints that addressed the protection and maintenance of communities or populations of benthic/epibenthic invertebrates, fish, birds, mammals, and reptiles (Table 3-1). These assessment endpoints were evaluated within a NBSA-specific risk analysis framework using the data and information collected under the Phase III RI/BERA sampling programs, and integrated site and ecosystem history information to inform the CSM and provide perspectives on potential changes in risk over time.

The potential for unacceptable risk was assessed using standard risk assessment practices and through development of LOEs to develop a WOE assessment for each receptor type, as described in Section 3. The LOEs were based on each measurement endpoint evaluated for the selected assessment endpoints, and associated strengths and uncertainties. In some cases, additional LOEs were available (e.g., site-specific toxicity data from the published literature) and were included in the WOE. The design of this risk assessment was focused on minimizing uncertainties, to the extent practicable, by being reasonably conservative in the exposure assumptions that were made and conducting sensitivity analyses using alternative assumptions for key input parameters to the exposure-risk models and calculations as described in Sections 6 through 9. In addition, conservative TRVs were generally used for all of the COPECs, and bounding of the potential effects was conducted by evaluating HQs calculated using both the NBSA TRVs and the TRVs developed by the USEPA (2014) for use in the LPR FFS. In addition, available site-specific ecotoxicity studies conducted in the NBSA were used for invertebrates, fish, and birds to quantify risks and/or inform the risk assessment.

The integrated summary of the risk findings for each AE are presented in Section 11.1. More detailed discussion can be found in the respective WOE evaluations for each receptor type, as presented above in Sections 6.4 (invertebrates), 7.6 (fish), 8.4 (birds), and 9.3 (mammals)[12]. The important sources of uncertainty in this BERA are summarized in Section 11.2. The integrated WOE risk findings and recommendations for risk drivers/final COCs are provided in Section 11.3. Final COCs selected for evaluation in the FS will be determined in consultation with USEPA.

~~12.1~~11.1 Integrated Risk Findings

A range of risk estimates were presented above in Sections 6 through 9 based on the various ME/LOE evaluated for each AE and, in some cases, based on multiple toxicity and/or exposure assumptions evaluated within each ME/LOE. In order to draw risk conclusions, strengths and weaknesses associated with each ME/LOE and set of assumptions were evaluated as described in Section 3.5.3, along with the overall indication of risk from multiple LOE. These integrated risk findings are presented in detail for benthic invertebrate communities in Table 6-17, epibenthic invertebrates in Table 6-21, fish in Table 7-24, birds in Table 8-15, and mammals in Table 9-8. A qualitative evaluation was conducted for reptiles. Table 11-1a provides a summary of these integrated risk characterization findings for epibenthic invertebrates

[12] A formal WOE evaluation was not conducted for reptiles. See Section 10 for the qualitative risk evaluation and LOE.

(softshell clam and blue crab), fish, birds, mammals, and reptiles and the AEs identified in Section 3; Table 11-1b provides this summary for benthic invertebrate communities (including polychaete worms) based on the results of the SQT evaluation.

As summarized in Table 11-1a, the Risk Conclusions regarding the potential for unacceptable risk for epibenthic invertebrates, fish, birds, and mammals fall into the following categories:

1. ~~Risk~~ Possible: LOAEL HQs ≥ 1 *and* sufficient and strong supporting LOEs. The COPECs associated with this Risk Conclusion are recommended as risk drivers/COCs.
2. Unlikely: LOAEL HQs ≥ 1 *and* sufficient and strong LOEs supporting a conclusion of no unacceptable risk. The COPECs associated with this Risk Conclusion are not recommended as risk drivers/COCs.
3. Uncertain: LOAEL HQs ≥ 1 *and* insufficient LOEs to draw a risk conclusion. While sufficient data were collected to evaluate ecological receptor exposures in the NBSA, toxicity data used to evaluate the exposures was selected primarily from published literature. Toxicity data for some receptors, such as tissue copper effects levels in fish, are limited in the literature. In cases where there is low confidence in the toxicity data and the various LOEs gave conflicting results, the available information was considered too uncertain to draw risk conclusions. The COPECs associated with this Risk Conclusion are recommended as COCs.
4. Not expected: LOAEL HQs < 1.

Overall, unacceptable risks to ~~fish and~~ wildlife (i.e., bird and mammal) receptors are either not expected or considered unlikely in the NBSA for all COPECs ~~except copper for white perch. The potential for unacceptable risk to white perch from copper in the NBSA is uncertain due to low confidence in both the dietary and tissue TRVs (see Section 7.7.4) and conflicting risk predictions based on the dietary and tissue HQs.~~ .

For fish, the potential for unacceptable risk was identified only for dioxins/furans and TEQ (2,3,7,8-TCDD, total dioxin/furan TEQ fish, dioxin/furan/PCB TEQ fish) for all fish receptors, total PCBs for forage fish in the Southwest Zone, and total DDx for white perch. Additionally, the potential for unacceptable risk to white perch from copper in the NBSA is highly uncertain due to low confidence in both the dietary and tissue TRVs (see Section 7.7.4),conflicting risk predictions based on the dietary and tissue HQs, and the fact that it is a physiologically regulated metal in fish.

For epibenthic invertebrates (i.e., softshell clam and blue crab), potential for unacceptable risks ~~are considered unlikely~~were identified for ~~most COPECs except~~ 2,3,7,8-TCDD ~~and Total~~, total PCBs, ~~for which unacceptable risk is considered possible~~arsenic, lead, and mercury for softshell clam ~~only. The~~ , and 2,3,7,8-TCDD, total PCBs, total DDx, mercury, and methylmercury for blue crab. For 2,3,7,8-TCDD and total PCBs, the finding of possible risk to softshell clam is due to tissue-based LOAEL HQs that exceed ~~1~~10 based on bivalve-specific TRVs and supporting site-specific toxicity data indicating potential reproductive impairment in oysters. The HQs based on bivalve-specific TRVs are considered less reliable for blue crab, but ~~not substantially enough (i.e., LOAEL HQ range of 3 to 6) to warrant a finding of likely~~were used to identify 2,3,7,8-TCDD and total PCBs as having potential for unacceptable risk. The finding of somewhat higher potential for unacceptable risk in softshell clams is consistent with the fact that these organisms are sessile throughout most of their life history, are particle filter feeders, and live in close association with the sediments.

For benthic invertebrate communities, the outcome of the SQT was used to draw risk conclusions, as summarized in Table 11-1b. For polychaete worms, supporting LOEs include tissue HQs and site-specific toxicity data, as presented in Table 11-1a. The benthic invertebrate (i.e., infaunal) communities that live within the sediments of the NBSA are not considered impacted, either from a species diversity/abundance/evenness standpoint, or with respect to the urban background communities that inhabit Jamaica Bay in the southern NY/NJ Harbor Estuary. Acute toxicity, based on 10-day amphipod survival was observed in only two of 30 locations. A total of six stations had 28-day survival significantly different from control (six other stations had 28-day survival less than 80% of control, but were not significant due to the variability in this endpoint). Eighteen samples had growth, another variable endpoint, that was significantly different than control but only seven samples had growth less than 50% of control. The SQT assessment and dose response modeling indicate localized areas of measured and predicted sediment toxicity, particularly in the southwest area of the NBSA (Figures 6-6 and 6-8). However, the ~~vast~~ majority of the sediments sampled throughout the NBSA are not acutely toxic and pose no incremental risk to benthic communities above urban reference conditions. The localized toxicity that does exist appears to be related to multiple COPECs that are all at or above the 90th percentile of their NBSA-wide concentration range in surface sediments at these locations.

~~12.2~~11.2 Summary of Key Uncertainties

A number of conservative practices were used and assumptions made for the risk assessment in this BERA, including:

- Wildlife representative receptor species were intentionally selected based on attributes (e.g., small foraging areas) that provide conservative estimates of exposure for other members of the feeding guild. Exposure parameters for the selected representative species from approved sources (e.g., USEPA Wildlife Exposure Factor Handbook [USEPA 1993]) were preferred sources of wildlife exposure information to reduce the uncertainty for the species living at this specific site.
- Prey tissue collected to evaluate avian dietary exposure was for species likely to be highly exposed (e.g., infaunal invertebrates) and assumed diets were based on these limited prey items.
- Exposure estimates assume that measured concentrations of COPECs in sediment and tissue are 100% bioaccessible via dietary uptake (dermal absorption via the gut). For many COPECs, this assumption overestimates exposure.
- Estimates of exposure assume that wildlife do not avoid contaminated areas or foods, and do not actively avoid areas of the site in close proximity to industrial development and/or uses.
- Exceedance of LOAEL-based effects thresholds developed based on exposure to individual fish, birds, or mammals is predictive of adverse population-level effects.

Specific uncertainties for each AE are discussed in detail in Sections 6.4 (invertebrates), 7.6 (fish), 8.3 (birds), and 9.2 (mammals). Because of the generally conservative approaches taken and other protective assumptions made or evaluated throughout this BERA, risk estimates are expected to be overestimated rather than underestimated.

~~12.3~~11.3 Weight of Evidence Conclusions and Risk Driver Recommendations

Based on the integrated risk management findings summarized in Tables 11-1a and 11-1b, unacceptable risk was considered possible for the following COPECs, which are recommended as risk drivers/final COCs:

- Dioxins/furans and TEQ (i.e., 2,3,7,8-TCDD, total dioxin/furan TEQ fish, dioxin/furan/PCB TEQ fish) – for ~~softshell clam~~shellfish (2,3,7,8-TCDD only), forage and predatory fish (Table 11-1a) and benthic communities (2,3,7,8-TCDD only; Table 11-1b)
- PCBs (i.e., total PCBs based on sum of 209 congeners) – for ~~softshell clam~~shellfish, and forage fish in the Southwest Zone (Table 11-1a) and benthic communities (Table 11-1b)
- Pesticides (total chlordane, total DDx [total of 2,4'- and 4,4'-DDT, DDE, and DDD], dieldrin, and hexachlorobenzene) ~~—~~for benthic communities (Table 11-1b), and total DDx for blue crab and white perch (Table 11-1a)
- PAHs (total PAH, HMW PAH, LMW PAH, alkylated PAH) – for benthic communities (Table 11-1b)
- Inorganics (arsenic, cadmium, chromium, copper, lead, mercury/methylmercury, nickel, silver, and zinc) – for benthic communities (Table 11-1b), arsenic, lead, mercury, and/or methylmercury for shellfish (Table 11-1a), and copper, based on uncertain risk to white perch (Table 11-1a).

As summarized in the bullets above, these COPECs ~~were selected~~are recommended as COCs primarily based on their potential local risk to benthic infaunal communities from the sediment toxicity endpoint in the SQT assessment. Dioxins/furans, PCBs, and ~~PCBs~~total DDx were also identified based on the WOE evaluation for ~~softshell clam~~shellfish and fish. Dioxins/furans, PCBs, total DDx, and mercury/methylmercury also showed some potential for risk in one or more of the tissue-based and/or dietary-based assessments for ~~fish,~~ birds~~,~~ and mammals, as summarized in Table 11-1a; however, unacceptable risk was ultimately considered unlikely in the NBSA for these COPECs and receptors based on the WOE.

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ATTACHMENT 1

Screening-Level Ecological Risk Assessment (SLERA) for the Newark Bay Study Area (Battelle 2008)

ATTACHMENT 2

Problem Formulation for the Newark Bay Study Area (Tierra 2013a)

ATTACHMENT 3

Newark Bay Study Area Reconnaissance Report (Tierra 2015a)

ATTACHMENT 4

Newark Bay Study Area Environmental Sample Collection Report (Normandeau 2017)

ATTACHMENT 5

Newark Bay Fish Sampling and Analysis, Amended Pathology Reports (EPL 2016a, 2016b)

ATTACHMENT 6

Chemical Residues in Cormorants from New York Harbor and Control Location (Parsons 2003)