REMEDIAL INVESTIGATION REPORT

REMEDIAL INVESTIGATION/FEASIBILITY STUDY, NEWTOWN CREEK

Prepared by
Anchor QEA, LLC
123 Tice Boulevard, Suite 205
Woodcliff Lake, New Jersey 07677

March 2023

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LIST OF ACRONYMS AND ABBREVIATIONS

Σ SEM – AVS	sum of simultaneously extracted metals minus acid volatile sulfide
μg/kg	micrograms per kilogram
μg/L	micrograms per liter
μmol/gram	micromoles per gram
2,3,7,8-TCDD	2,3,7,8-tetrachlorodibenzo-p-dioxin
95% UCL	95% upper confidence limit of the mean
ADCP	acoustic Doppler current profiler
Ag	silver
Amoco	American Oil Company
Amtrak	National Railroad Passenger Corporation
AOC	Administrative Order on Consent
As	arsenic
ASAC	activity-specific acceptance criteria
AST	aboveground storage tank
ASTM	ASTM International
ATSDR	Agency for Toxic Substances and Disease Registry
AVS	acid volatile sulfide
Ba	barium
BAPS	Borden Avenue Pumping Station
Bayside Fuel Oil Depot	Bayside Fuel Oil Depot – 1100 Grand St (DAR No. 51)
B.C.F. Oil	B.C.F. Oil Refining, Inc. (DAR No. 27)
BERA	*Baseline Ecological Risk Assessment*
Be	beryllium
BHHRA	*Baseline Human Health Risk Assessment*
BMP	best management practice
BSAF	biota-sediment accumulation factor
BTEX	benzene, toluene, ethylbenzene, and xylene
Ca	calcium
CARP	Contamination Assessment and Reduction Project
CBP	average number of chlorine atoms per biphenyl molecule
CBR	critical body residue

Cd	cadmium
CDM	Collective Data Method
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CFR	Code of Federal Regulations
CFT	contaminant fate and transport
cm	centimeter
CM	creek mile
cm/day	centimeters per day
cm/year	centimeters per year
Con Edison	Consolidated Edison Company of New York
COPC	contaminant of potential concern
COPEC	contaminant of potential ecological concern
Cr	chromium
CRR	New York Codes, Rules, and Regulations
Cs-137	cesium-137
C19-C36	C19-C36 aliphatic hydrocarbons
CSM	conceptual site model
CSO	combined sewer overflow
CTE	central tendency exposure
Cu	copper
CV	coefficient of variation
CVOC	chlorinated volatile organic compound
CWA	Clean Water Act
d/b/a	doing business as
DAR	*Data Applicability Report*
DMR	discharge monitoring report
DO	dissolved oxygen
DOC	dissolved organic carbon
DQO	data quality objective
DRO	C10-C28 diesel range organics
DSR	Data Summary Report
EDD	electronic data deliverable

EMC	event mean concentration
ENR	enhanced natural recovery
EPI Suite	Estimation Program Interface Suite
ExxonMobil	Exxon Mobil Oil Corporation
Fe	iron
feet^3/day	cubic feet per day
FES	field gas ebullition survey
FMRM	*Final Modeling Results Memorandum*
f_{OC}	fraction organic carbon
FoD	frequency of detection
FS	Feasibility Study
FS DSR Part 1	*Feasibility Study Data Summary Report Part 1*
FS FP Work Plan	*Feasibility Study Field Program Work Plan*
FS FSAP	*Feasibility Study Field Sampling and Analysis Plan*
FS Gas Ebullition DER	*Feasibility Study Gas Ebullition Data Evaluation Report*
FS Geotechnical DER	*Feasibility Study Geotechnical Data Evaluation Report*
FS NAPL DER	*Feasibility Study Nonaqueous Phase Liquid Mobility Data Evaluation Report*
FS QAPP	*Feasibility Study Quality Assurance Project Plan*
FSAP	*Field Sampling and Analysis Plan*
FSZ	fish sampling zone
G	acceleration due to gravity
g/cm^3	gram per cubic centimeter
GAC	granular activated carbon
geo-neutral	geographically neutral
Getty Terminals	Getty Terminals Corp. #58220 (DAR No. 47)
HEAST	*Health Effects Assessment Summary Table*
Hg	mercury
HI	hazard index
HLC	Henry's Law Constant
HPAH	high-molecular-weight polycyclic aromatic hydrocarbon
HQ	hazard quotient

Hugo Neu	Hugo Neu Schnitzer (aka SIMS Hugo Neu) (DAR No. 125)
IRIS	Integrated Risk Information System
IRM	Interim Remedial Measure
IUR	Inhalation Unit Risk
J/mol	joules per mole
K	potassium
K_d	partition coefficient
kg	kilogram
kg/year	kilograms per year
KM	Kaplan-Meier
K_{OC}	organic carbon partition coefficient
K_{OW}	octanol/water partition coefficient
L/kg	liters per kilogram
L^3/T	volume per unit time
LCL	lower confidence limit
LiDAR	Light Detection and Ranging
LIRR	Long Island Rail Road
LNAPL	light nonaqueous phase liquid
LOE	line of evidence
LOEC	lowest observed effect concentration
LPAH	low-molecular-weight polycyclic aromatic hydrocarbon
LTCP	long-term control plan
M/L^3	mass per unit volume
M/T	mass per unit time
MAM2	*Modeling Approach Memorandum (2)*
MAM3	*Modeling Approach Memorandum (3)*
MDAC	minimum data acceptance criteria
MDL	method detection limit
Mg	magnesium
MG	million gallons
$mg/cm^2/day$	milligrams per square centimeters per day
mg/kg	milligrams per kilogram
mg/kg-day	milligrams per kilograms per day

mg/L	milligrams per liter
MGD	million gallons per day
MGP	manufactured gas plant
MGY	million gallons per year
mL	milliliter
MLLW	mean lower low water
MLW	mean low water
MNR	monitored natural recovery
Mobil	Mobil Oil Corporation
Morgan Oil	Morgan Oil Terminal, Brooklyn (DAR No. 60)
Motiva	Motiva Enterprises, LLC
MPN	Most Probable Number
MRL	Minimal Risk Level
MS4	municipal separate storm sewer system
MSGP	multisector general permit
MT/year	metric tons per year
MTA	Metropolitan Transportation Authority
Na	sodium
NA	not available
NAPL	nonaqueous phase liquid
NAVD88	North American Vertical Datum of 1988
NCG	Newtown Creek Group
NCP	*National Oil and Hazardous Substances Pollution Contingency Plan*
ng/kg	nanograms per kilogram
ng/L	nanograms per liter
Ni	nickel
NJADN	New Jersey Atmospheric Deposition Network
NPDES	National Pollutant Discharge Elimination System
NSR	net sedimentation rate
NTU	nephelometric turbidity units
NY/NJ	New York/New Jersey
NYC	New York City
NYCDEP	New York City Department of Environmental Protection

NYCDOS	New York City Department of Sanitation
NYSDEC	New York State Department of Environmental Conservation
NYSDOH	New York State Department of Health
OC	organic carbon
OHW	ordinary high water
OIU	open industrial use
OPA	oil-particle aggregate
p	The smallest value of α at which the null hypothesis would be rejected for the given observations. The p value of the test is sometimes called the critical level, or the significance level, of the test.
PAH	polycyclic aromatic hydrocarbon
Pb	lead
Pb-210	lead-210
PCB	polychlorinated biphenyl
PDRC	Phelps Dodge Refining Corporation
Phase 1 DSR Submittal No. 1	*Phase 1 Remedial Investigation Field Program Data Summary Report – Submittal No. 1*
Phase 1 DSR Submittal No. 3	*Phase 1 Remedial Investigation Field Program Data Summary Report – Submittal No. 3*
Phase 1 IDR	*Phase 1 Remedial Investigation Interim Data Report*
Phase 2 DSR	*Phase 2 Remedial Investigation Field Program Data Summary Report*
Phase 2 FSAP Volume 1	*Phase 2 Field Sampling and Analysis Plan – Volume 1*
Phase 2 FSAP Volume 2	*Phase 2 Field Sampling and Analysis Plan – Volume 2*
Phase 2 QAPP	*Phase 2 Quality Assurance Project Plan*
Phase 2 RI Work Plan Volume 1	*Phase 2 Remedial Investigation Work Plan – Volume 1*
Phase 2 RI Work Plan Volume 2	*Phase 2 Remedial Investigation Work Plan – Volume 2*
PM&F	plastics molding and forming industry

POC	particulate organic carbon
POW	Pratt Oil Works
PPRTV	Provisional Peer Reviewed Toxicity Value
propwash	propeller wash
psu	practical salinity unit
Pv	total pore volume
Q1	Quarter 1
Q2	Quarter 2
Q3	Quarter 3
QAPP	*Quality Assurance Project Plan*
Quanta/RAD II	Quanta Resources a/k/a Review Ave. Development II (DAR No. 39)
r	correlation coefficient
r^2	coefficient of determination
RAD I	Review Avenue Development I (DAR No. 41)
REMAP	Regional Environmental Monitoring and Assessment Program
RfC	reference concentration
RfD	reference dose
RHA	Rivers and Harbors Act
RI	Remedial Investigation
RI Report	*Remedial Investigation Report*
RI/FS	Remedial Investigation/Feasibility Study
RI/FS Work Plan	*Remedial Investigation/Feasibility Study Work Plan*
RL	reporting limit
RME	reasonable maximum exposure
RSL	Regional Screening Level
RUM	Representative and Unique (Stormwater) Method
Sd	standard deviation
SEM	simultaneously extracted metals
SF	slope factor
Shell	Shell Oil Company
SL	screening level
SLERA	screening level ecological risk assessment

SMIA	Significant Maritime and Industrial Area
SOCONY	Standard Oil Company of New York
SOM	Specific Outfall Method
SPDES	State Pollutant Discharge Elimination System
SPME	solid-phase microextraction
SQT	sediment quality triad
SSAM	*Sources Sampling Approach Memorandum*
sVGP	Small Vessel General Permit
SVOC	semivolatile organic compound
TBD	to be determined
TDI	total daily intake
TDS	total dissolved solids
TEQ	toxic equivalence quotient
Tidewater	Tidewater Oil Company
TIR	thermal infrared
TOC	total organic carbon
total DDx	2,4′ and 4,4′-DDD, -DDE, -DDT
TPAH (17)	total polycyclic aromatic hydrocarbon (17)
TPAH (34)	total polycyclic aromatic hydrocarbon (34)
TPCB	total polychlorinated biphenyl
TPH	C9-C40 total petroleum hydrocarbon
TRV	toxicity reference value
TSCA	Toxic Substances Control Act
TSS	total suspended solids
TU	toxic unit
UGA	Upper Glacial Aquifer
USACE	U.S. Army Corps of Engineers
USCG	U.S. Coast Guard
USEPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey
UST	underground storage tank
UV	ultraviolet
VOC	volatile organic compound

WBI	Weisberg Biotic Index
WOE	weight-of-evidence
WPA	*Phase 1 Remedial Investigation Work Plan Addendum*
WPCP	Water Pollution Control Plant
wt%	weight percent
WWTP	wastewater treatment plant
Zn	zinc

EXECUTIVE SUMMARY

Introduction

The Newtown Creek Superfund Site was added to the National Priorities List and published in the Federal Register on September 29, 2010. This *Remedial Investigation Report* (RI Report) presents the results of a comprehensive investigation conducted between 2011 and 2018, designed to characterize the Study Area and to assess potential risks to human health and the environment. This RI Report presents the results of the investigation and, together with the *Baseline Human Health Risk Assessment* (BHHRA; Appendix H) and the *Baseline Ecological Risk Assessment* (BERA; Appendix I), provides the foundation for evaluating remedial alternatives during the Feasibility Study (FS).

The Newtown Creek Remedial Investigation (RI) data collection program was conducted in two phases, which are referred to as Phase 1 and Phase 2 throughout the document, followed by Part 1 of the FS field program. All studies have followed methods and procedures described in U.S. Environmental Protection Agency (USEPA)-approved work plans and conducted directly under USEPA oversight. Specifically, these studies focused on the following objectives:

- Phase 1 sampling: Intended to broadly characterize chemical and physical features of the Study Area.
- Phase 2 sampling: Conducted to fill data gaps and collect additional data needed to support the risk assessments and modeling, as well as the point sources, nonaqueous phase liquid (NAPL), and groundwater evaluations.
- Part 1 of the FS field program: Conducted to collect additional data to support the development and evaluation of remedial alternatives; these data are presented in this RI Report.

Additional FS field program studies (i.e., Part 2 of the FS Field Program) are presented in the *Feasibility Study Field Sampling Program Data Summary Report Part 2* (Anchor QEA 2020a).

In addition to the field sampling and surveys, the Remedial Investigation/Feasibility Study (RI/FS) includes a modeling effort consisting of five components: hydrodynamics, sediment transport, groundwater, chemical fate and transport (CFT), and bioaccumulation.[1] These models are in various phases of development and will be used to evaluate remedial alternatives in the FS.

Site Setting and Physical Characteristics

Newtown Creek forms part of the border between the boroughs of Brooklyn and Queens, New York City, New York. It is a tidal inlet to the East River with no natural tributary inflows. It is approximately 3.8 miles long and comprises a main channel and five tributaries (Dutch Kills, Maspeth Creek, Whale Creek, East Branch, and English Kills). A navigation channel extends through the main stem and into portions of Whale Creek and English Kills. The average width of the main stem is approximately 100 meters, and the average depth ranges from approximately 5 to 6 meters, depending on location. All five tributaries tend to be narrower and shallower than the main channel; average widths range from approximately 50 to 70 meters, and average depths range from less than 1 meter to 5 meters. The

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[1] On September 28, 2021, USEPA sent an email to the Newtown Creek Group (NCG) stating that the development of the bioaccumulation model should be discontinued. Based on USEPA's internal review; discussions with the NCG, New York City Department of Environmental Protection (NYCDEP), and the stakeholder group; and consultation with USEPA's Contaminated Sediments Technical Assistance Group, the USEPA concluded that finalizing the bioaccumulation model would not have a sufficiently beneficial outcome for the project to warrant the significant time and effort that would be required to complete it. USEPA concluded that a plan for communication of remedial expectations could be based around the monitoring program and the empirical sediment and porewater concentrations, empirical biota tissue concentrations, and predictions from the CFT model. The bioaccumulation model was discontinued because USEPA concluded: 1) the model was unlikely to advance the development of preliminary remediation goals; 2) the model was not necessary for evaluating alternatives as part of the FS process because the CFT model would be used to evaluate the relative magnitude of differences between remedial alternatives, and empirical monitoring of biota tissue concentrations will assess how the site responds to tissue-based risk thresholds; and 3) the model would be subject to particularly high levels of uncertainty at the Newtown Creek site due to the combination of migratory exposure of key organisms and uncertainty regarding off-site exposures. Because the off-site exposure zone is not a part of the hydrodynamic, sediment transport, and CFT models, it was unlikely that the bioaccumulation model would have provided accurate forecast results.

Administrative Order on Consent (AOC) defines the Study Area as Newtown Creek and the five tributaries extending up to the ordinary high water mark.[2,3]

The land use around Newtown Creek from the 1800s through the present has been predominately industrial. This industrial development occurred in parallel with municipal use of Newtown Creek as a receiving waterbody of both stormwater and wastewater discharges. Newtown Creek continues to be a major receiving waterbody of industrial and municipal separate storm sewer system (MS4) discharges and combined sewer overflow (CSO) discharges (containing combined flows of stormwater, sanitary wastewater, and industrial wastewater), as well as treated effluent from the Newtown Creek wastewater treatment plant (WWTP) overflow during rainfall events. It is also a designated Significant Maritime and Industrial Area (SMIA), which will continue to give preference to commercial use of the creek and industrial uses in upland areas. Modifications to Newtown Creek, such as fill placement and bulkheading along shorelines that have occurred over time, have resulted in a system that is largely adapted for industrial, municipal, and navigational purposes. Consequently, the land use history and urban landscape in which Newtown Creek exists shapes the conceptual site model and informs the nature and extent of contaminants of potential concern (COPCs) and potentially significant sources, as well as key fate and transport characteristics, pathways, and exposure scenarios.

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[2] The Newtown Creek Superfund Site Study Area is described in the AOC as encompassing the body of water known as Newtown Creek, situated at the border of the boroughs of Brooklyn (Kings County) and Queens (Queens County) in the City of New York and the State of New York, roughly centered at the geographic coordinates of 40° 42' 54.69" north latitude (40.715192°) and 73° 55' 50.74" west longitude (-73.930762°), having an approximate 3.8-mile reach, including Newtown Creek proper and its five branches (or tributaries) known respectively as Dutch Kills, Maspeth Creek, Whale Creek, East Branch, and English Kills, as well as the sediments below the water and the water column above the sediments, up to and including the landward edge of the shoreline, and including also any bulkheads or riprap containing the waterbody, except where no bulkhead or riprap exists, then the Study Area shall extend to the ordinary high water mark, as defined in 33 Code of Federal Regulations § 328(e) and the areal extent of the contamination from such area, but not including upland areas beyond the landward edge of the shoreline (notwithstanding that such upland areas may subsequently be identified as sources of contamination to the waterbody and its sediments or that such upland areas may be included within the scope of the Newtown Creek Superfund Site as listed pursuant to Section 105(a)(8) of Comprehensive Environmental Response, Compensation, and Liability Act [CERCLA]).

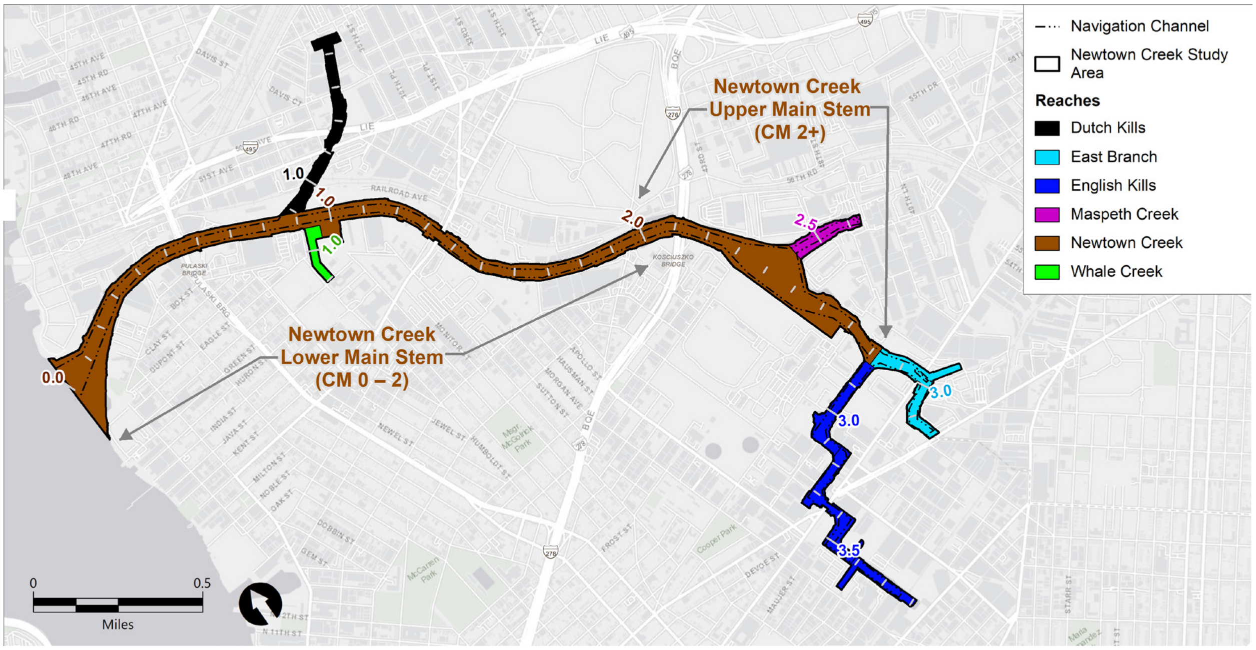
[3] The term "creek" is used interchangeably with "Study Area" throughout this RI Report.

The natural hydrodynamics of the Study Area are dominated by twice-daily tidal exchange with the East River and by rainfall-related flows from point sources and overland flow. Tidal mixing with East River water is most pronounced in creek mile (CM) 0 – 2 of the main stem, but continues to a significant degree beyond CM 2. Suspended solids are introduced into the Study Area water column primarily by the twice-daily tidal inflows from the East River and from the following: periodic discharges from CSO, MS4, and other point source stormwater discharges; overland stormwater flow; and the Whale Creek WWTP treated effluent overflow. These solids are transported and mixed within the surface water, and a portion of them eventually settle, continuously adding to, covering, and mixing with the existing sediment bed. The sediment bed throughout Newtown Creek is a cohesive (muddy) bed that is primarily net depositional, due to the low near-bed current velocities. Hydrodynamic processes (i.e., tidal currents and density-driven circulation) generate relatively low, near-bed current velocities throughout large portions of the Study Area, which result in minimal or near-zero erosion of the sediment bed, except in areas where vessel traffic may cause periodic scouring of the bed, or in shallow areas (i.e., sediment mounds) near large CSO outfalls where high current velocities can occur as a result of point source discharges during wet weather events (e.g., see Section 5.3.2.2 of Appendix G).

Based on some of the unique, site-specific Study Area characteristics noted previously, Newtown Creek is evaluated in this RI in the following three primary reaches (Graphic ES-1):

- **The lower main stem, from the mouth to approximately CM 2 (CM 0 – 2)**
  - CM 0 – 2 is characterized by extensive tidal exchange with the East River. Depositing solids originate primarily from the East River.
- **The upper main stem, including the Turning Basin (CM 2+)**
  - CM 2+ is a more complex portion of the Study Area. Depositing solids originate both from downstream (the East River) and upstream (primarily CSO and stormwater outfalls). Depositional characteristics within CM 2+ vary relative to position of the navigational channel, influences of vessel traffic, and shoreline features.

- **The tributaries**
  - The tributaries exhibit low surface water current velocities under typical conditions. CSO and storm-related point source inflows provide nearly all the solids that deposit on the sediment bed in the upper tributaries (i.e., Maspeth Creek, East Branch, and English Kills). Each tributary differs in circulation, deposition characteristics, and solids sources.



Graphic ES-1. Study Area Reaches

Nature and Extent of Contamination and Fate and Transport Characteristics

A primary focus of the RI field program was to delineate the nature and extent of contamination in the Study Area. Based on the results from the BERA and BHHRA, contaminants that were found to contribute to human health or ecological risk were used to characterize the nature and extent of contamination in the RI Report. These contaminants are as follows: total polycyclic aromatic hydrocarbon (17) (TPAH [17]),[4] total polycyclic

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[4] This includes the 16 USEPA priority pollutant polycyclic aromatic hydrocarbons (PAHs), as well as 2-methylnaphthalene.

aromatic hydrocarbon (34) (TPAH [34]),[5] C19-C36 aliphatic hydrocarbons (C19-C36),[6] total polychlorinated biphenyl (TPCB),[7] 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD),[8] copper (Cu), lead (Pb), and dieldrin.

While these eight contaminants (or groups of contaminants) were used to characterize nature and extent of contamination, the degree to which they contribute to human health and ecological risks varies.[9] Hydrocarbons, TPCB, and Cu contribute to risk in the human health and/or ecological risk assessments and represent three broad classes of contaminants. Hydrocarbons include TPAH (17), TPAH (34), and C19-C36 (as well as other compounds), and each of these sums was found to contribute to ecological risk (to varying degrees). 2,3,7,8-TCDD and Pb also contribute to risk in the human health and/or ecological risk assessments, albeit to a lesser degree than TPCB and Cu, respectively. Dieldrin was also

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[5] This includes both the 17 compounds in TPAH (17), as well as 17 other C1- to C4-alkylated homologs of 2- to 6-ring PAHs.

[6] This is representative of hydrocarbons having between 19 and 36 carbon atoms and is also a component of C9-C40 total petroleum hydrocarbons (TPH, which also includes the C10-C28 diesel range organics [DRO]).

[7] This includes 209 individual chlorinated compounds or congeners consisting of a biphenyl molecule and one to ten chlorine atoms.

[8] This compound is a major contributor to the total dioxin/furan toxic equivalence quotient (TEQ).

[9] TPAH (17), TPAH (34), and C19-C36 are primary risk drivers in the BERA (see Appendix I), TPCB is a primary risk driver in the BHHRA and BERA (see Appendices H and I, respectively), and Cu was selected as a representative metal because of some potential ecological risk, and bulk sediment concentrations are elevated relative to screening benchmarks in sediment in CM 2+. 2,3,7,8-TCDD was identified as a risk driver in the BHHRA and as a contaminant of potential ecological concern (COPEC) in the BERA, and Pb was identified as a COPEC in the BERA. Although dieldrin was not identified as a COPC or COPEC, it was included in the nature and extent evaluation of surface sediment and tissue because of elevated concentrations in polychaete tissue in one reach of the Study Area (English Kills). These eight contaminants (or groups of contaminants) were used to characterize nature and extent of contamination; however, not all of these contaminants were included for evaluations of sources and fate and transport, because: 1) the distributions in environmental media (including surface sediment) are broadly similar to those within the same class (i.e., hydrocarbons, bioaccumulative organics, and metals); 2) in some of the locations or some of the media, some of these contaminants (e.g., C19-C36, 2,3,7,8-TCDD, and Pb) were either not analyzed or were infrequently detected (in the case of surface water, porewater, and groundwater—these contaminants were detected at generally high frequencies in sediment); and 3) their fate and transport characteristics (i.e., partitioning behavior) are similar, especially to others within the same class. As such, TPAH (17), TPCB, and Cu were used for the evaluations of sources, fate and transport, and the quantitative aspects of the conceptual site model in the RI Report. However, going forward into the FS, additional contaminants will continue to be considered; for example, due to differences in hydrocarbon distributions, TPAH (17) cannot be considered a surrogate for other hydrocarbons, so the FS will consider the other hydrocarbon groups (i.e., TPAH [34] and C19-C36) individually.

evaluated for nature and extent in some media because elevated concentrations were observed in benthic invertebrate tissue in one portion of the Study Area (i.e., English Kills).

The distribution of these contaminants in the surface sediment (defined operationally as a depth of 0 to 15 centimeters [cm; 0 to 6 inches]), subsurface sediment (from 15 cm [6 inches] depth to the interface with the underlying native material), native material, surface water, and NAPL in the Study Area are summarized in the following sections.[10]

Sediment

TPAH (17), TPAH (34), C19-C36, TPCB, 2,3,7,8-TCDD, Cu, and Pb concentrations in surface sediment are summarized in the following graphics (Graphics ES-2 through ES-8, respectively).[11] In these graphics, the main stem of Newtown Creek extends from the mouth of the creek at the East River upstream through the Turning Basin. The main stem is divided into three segments: CM 0 – 1, CM 1 – 2 (shown as one reach in Graphic ES-1), and CM 2+. Each tributary is represented individually. These graphics also show the surface sediment data from reference areas for comparison. These reference areas were selected by USEPA to evaluate physical, chemical, and biological conditions in waterbodies that span four categories of industrial development and influence from CSO discharges, specifically Industrial/CSO, Industrial/Non-CSO, Non-Industrial/CSO, and Non-Industrial/Non-CSO.

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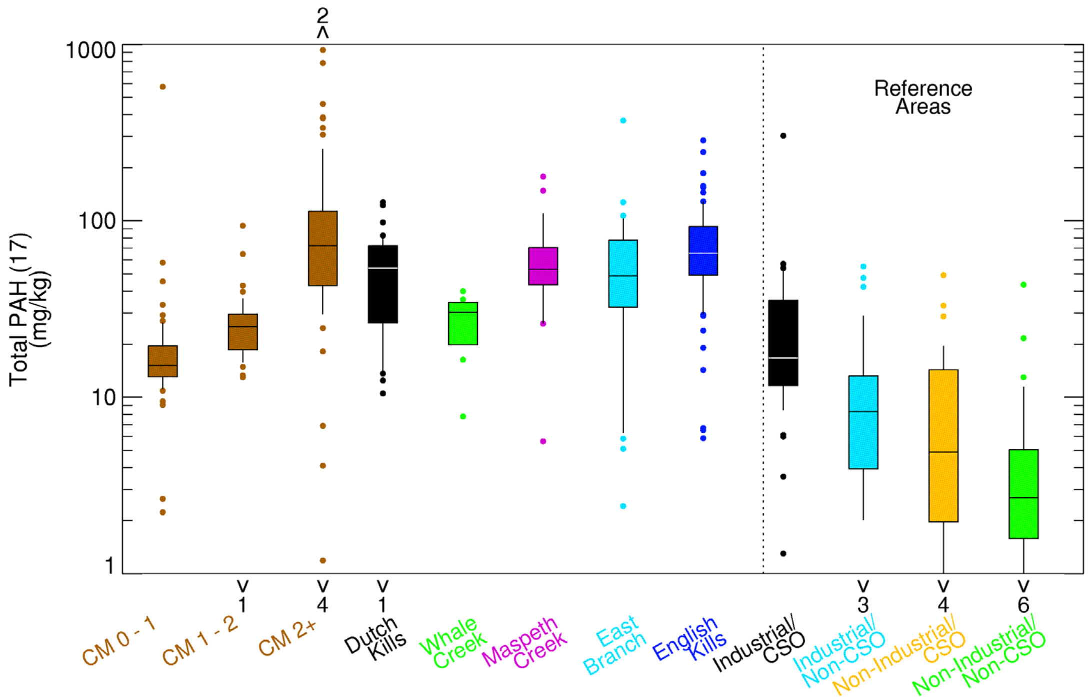
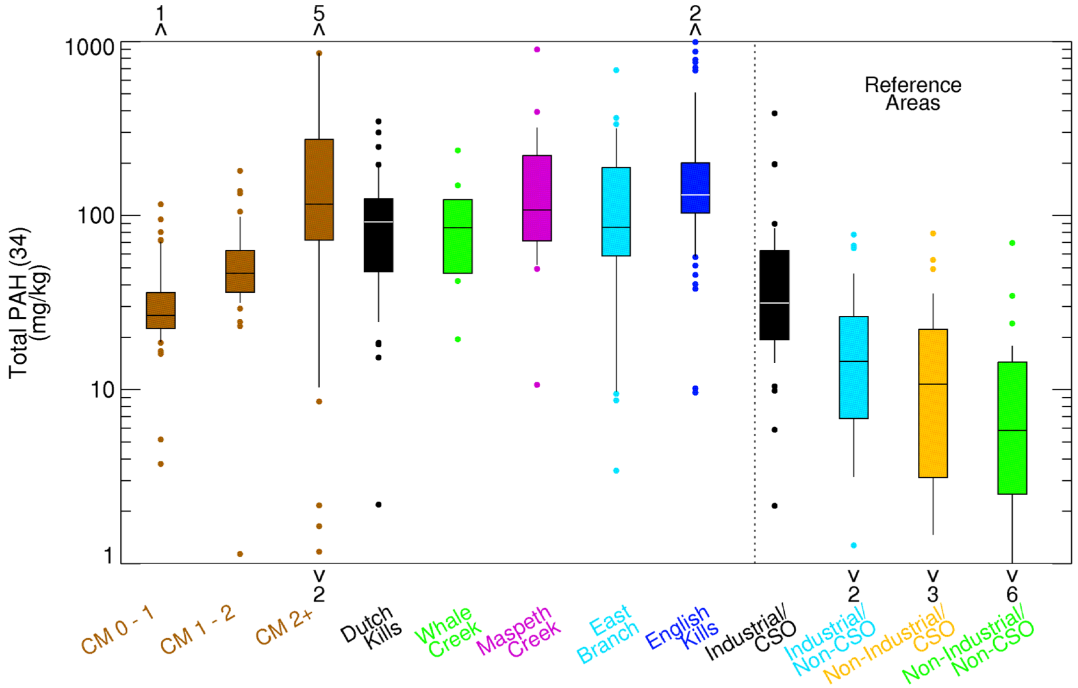
[10] Dieldrin was excluded from these sections because it was not identified as a COPC or COPEC in the risk assessments.

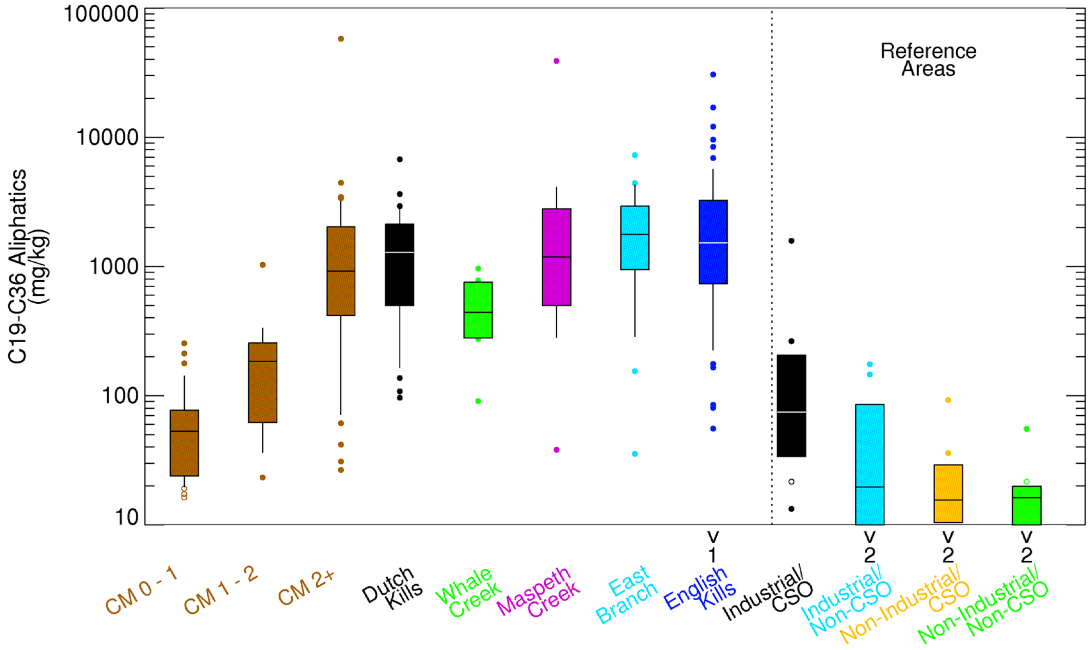
[11] In Graphics ES-2 through ES-8, the boxes represent the 25th and 75th percentiles of the data, and the vertical lines represent the 10th and 90th percentiles. The horizontal line through each box represents the median. All values lying outside the 10th and 90th percentiles are indicated individually. The caret symbols represent individual values that are above or below the panel; the number of values outside the panel is also indicated. Surface sediment includes data collected within the top 15 cm (6 inches) of the sediment bed.



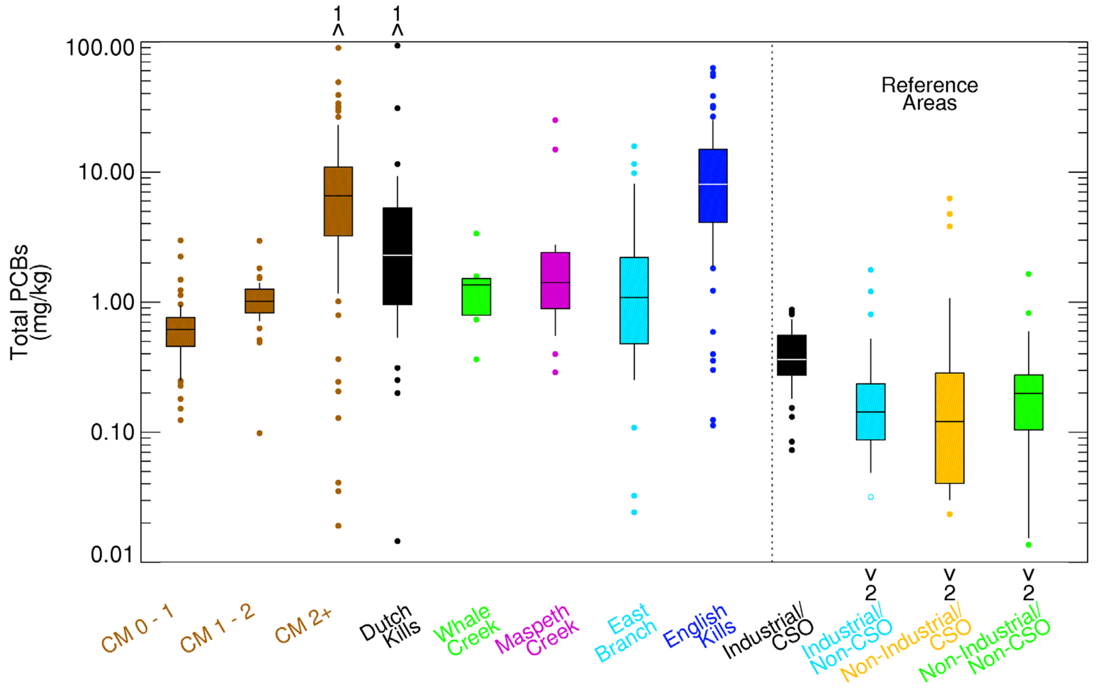
Graphic ES-2. TPAH (17) in Surface Sediment



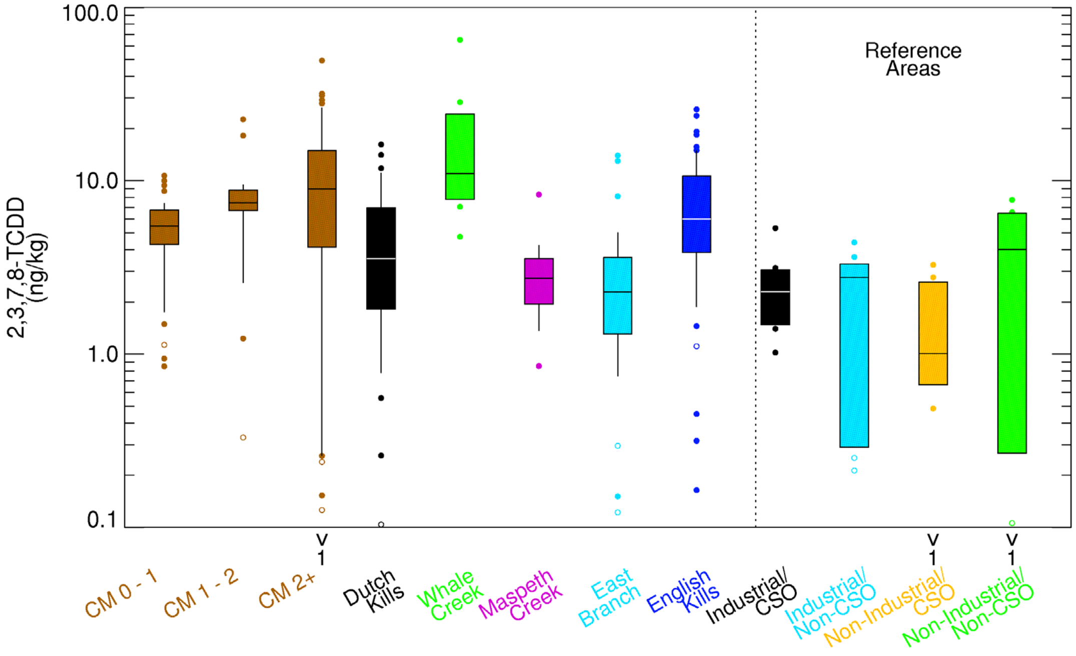
Graphic ES-3. TPAH (34) in Surface Sediment



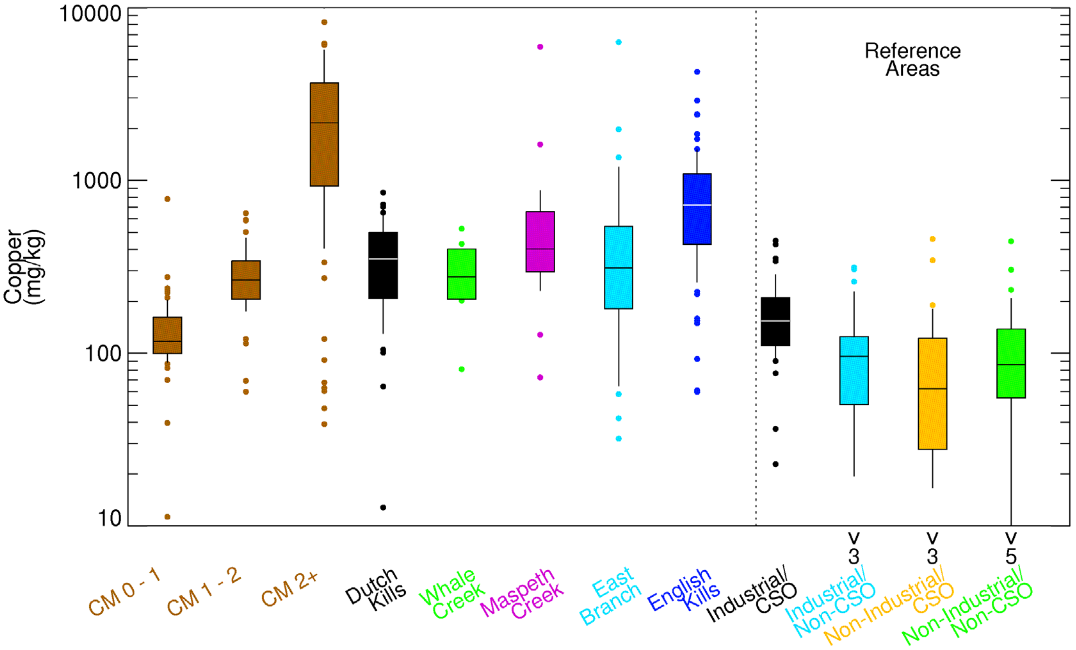
Graphic ES-4. C19-C36 in Surface Sediment



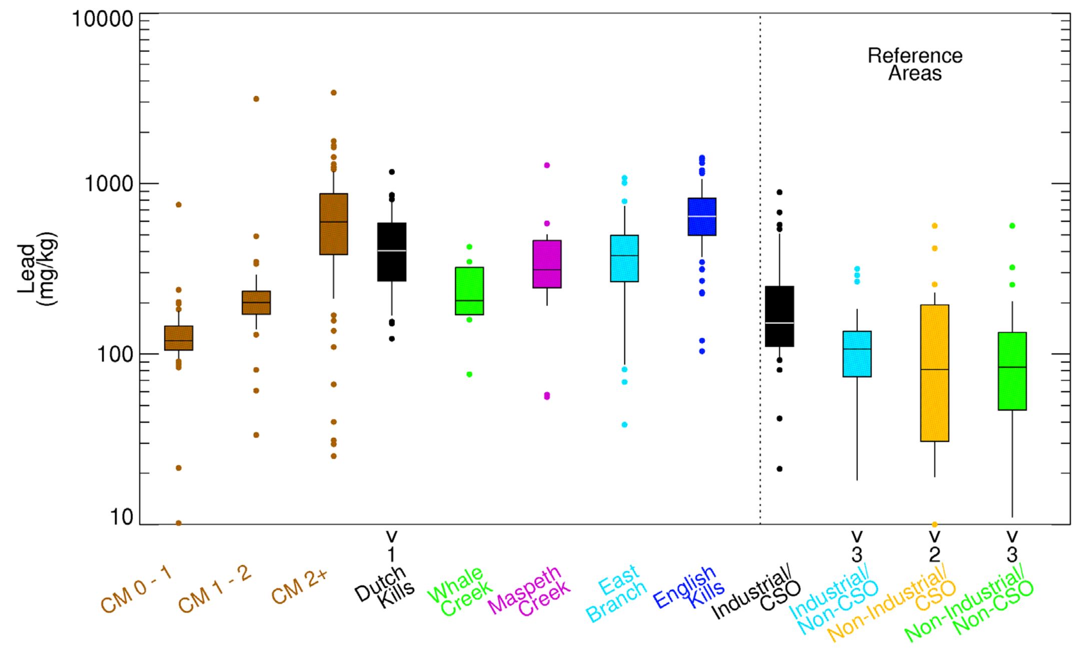
Graphic ES-5. TPCB in Surface Sediment



Graphic ES-6. 2,3,7,8-TCDD in Surface Sediment



Graphic ES-7. Copper in Surface Sediment



Graphic ES-8. Lead in Surface Sediment

Notable patterns in the surface sediment data are as follows:

- **Surface sediment, CM 0 – 2.** Surface sediment TPAH (17), TPAH (34), C19-C36, TPCB, 2,3,7,8-TCDD, Cu, and Pb concentrations in CM 0 – 1 are generally the lowest in the Study Area and are consistent with reference areas,[12] as represented by the Industrial/CSO reference area data and in the case of 2,3,7,8-TCDD all reference area categories. Concentrations in CM 1 – 2 are higher than those in CM 0 – 1, but also are generally consistent with (or approaching) reference area concentrations in the Industrial/CSO reference areas and in the case of 2,3,7,8-TCDD all reference area categories.
- **Surface sediment, CM 2+.** The highest surface sediment concentrations for TPAH (17), TPAH (34), C19-C36, TPCB, 2,3,7,8-TCDD, Cu, and Pb in the main stem are observed in CM 2+, with most values being above reference area concentrations.
- **Surface sediment, tributaries.** Concentrations in tributaries are generally higher than in CM 0 – 2 and generally exceed reference area concentrations as a result of the mixing of ongoing sources with residual historical contamination. The highest TPAH (17), TPAH

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[12] Concentrations in surface sediment samples collected from reference areas are located throughout the New York Harbor and Jamaica Bay area and are considered representative of reference area sediment concentrations.

(34), Cu, and Pb tributary concentrations, as well as elevated C19-C36, TPCB, and 2,3,7,8-TCDD concentrations, are observed primarily in the lower 0.5 mile of English Kills (see Section 4.2.3). The highest C19-C36 concentrations are observed throughout English Kills (with elevated concentrations at multiple locations in the other tributaries as well). The highest TPCB and 2,3,7,8-TCDD concentrations are observed in Dutch Kills and Whale Creek, respectively. In some tributaries, specifically East Branch and English Kills, concentrations decrease moving upstream, toward the head of each tributary (see Section 4.2.3).

In subsurface sediment, TPAH (17), TPAH (34), C19-C36, TPCB, 2,3,7,8-TCDD, Cu, and Pb concentrations are higher than in surface sediment in nearly all cases throughout the Study Area. The only exceptions are C19-C36 in Dutch Kills and East Branch and 2,3,7,8-TCDD in Dutch Kills, where surface and subsurface concentrations are generally similar. Like surface sediment, subsurface sediment concentrations in CM 0 – 2 are generally the lowest near the mouth of the Study Area and increase moving upstream, with the highest subsurface sediment concentrations in the main stem being observed in CM 2+. Subsurface sediment concentrations generally increase with depth, reaching a peak several feet below the mudline or increasing until native material is reached. Elevated contaminant concentrations generally are not present in the native material.

The subsurface sediment appears relatively stable. This is supported by the following:

- Lower concentrations of COPCs in surface sediment, as compared to subsurface sediment, throughout the Study Area
- Low current velocities throughout the Study Area that result in minimal or no erosion of the sediment bed, except in localized areas owing to propeller wash disturbance and in areas near point sources that discharge during wet weather events
- Net depositional sediment bed throughout the Study Area (deposition rate varies by location), based on multiple lines of evidence (LOEs), including sediment radioisotope studies, bathymetric surveys, and historical dredging records
- Pre- and post-Hurricane Sandy bathymetric surveys, which indicate minimal erosion of the sediment bed during the anomalous current velocities generated by the storm surge

Surface Water

In general, surface water contaminant concentrations exhibit considerably less spatial gradients than surface sediment. This limited spatial pattern is primarily due to mixing and to the influence of the East River. In general, wet weather concentrations were greater than dry weather concentrations, indicating the importance of ongoing point sources and stormwater-related events occurring in the Study Area.

Notable patterns in the data are as follows:

- **Dry weather, CM 0 – 2 and CM 2+.** TPAH (17), TPAH (34), TPCB, Cu, and Pb concentrations generally show little overall gradient in the main stem. Concentrations throughout the main stem are generally within the range of the East River, with increases moving upstream in some cases (e.g., TPCB and TPAH [34] in CM 1 – 2 and CM 2+). 2,3,7,8-TCDD was only detected in one sample from CM 1 – 2. C19-C36 was not analyzed in surface water samples.
- **Dry weather, tributaries.** The highest dry weather TPAH (17), TPAH (34), and TPCB concentrations are observed in English Kills and to a lesser extent in East Branch. Concentrations in the other tributaries are generally similar to one another and are consistent with those observed in the main stem and in the East River. Dry weather Cu and Pb concentrations are similar across all the tributaries and are generally consistent with those observed in the main stem and in the East River. 2,3,7,8-TCDD was not detected in dry weather surface water samples from the tributaries.
- **Wet weather.** In the main stem, wet weather concentrations of TPAH (17), TPAH (34), TPCB, Cu, and Pb increase somewhat with distance upstream. These patterns suggest influence from CSOs, other point sources, and overland flow. 2,3,7,8-TCDD was only detected in one wet weather sample from CM 0 – 1 and two wet weather samples from CM 1 – 2. Wet weather TPAH (17), TPAH (34), TPCB, and Cu concentrations are higher than the corresponding dry weather concentrations in all reaches of the Study Area. Comparisons between wet and dry weather Pb concentrations are confounded by varying detection limits and low frequency of detection in dry weather samples.

NAPL

The presence and extent of NAPL were extensively investigated during the RI and FS Part 1 field programs. Multiple field investigations and the collection of hundreds of surface sediment grabs and cores were used to evaluate NAPL presence and extent in Study Area sediment and native material. As shown in Graphic ES-9, NAPL presence or absence was identified using a two-part process combining direct visual observation of sediment and native material in the cores, along with the performance of shake tests and visually observing if NAPL separated from the sediment or native material.[13] The presence of NAPL blebs or a NAPL layer in a shake test indicates that NAPL is present. The lack of NAPL blebs or a NAPL layer in a shake test (i.e., no observation, or sheen only) confirms that NAPL is not present, as indicated in Note 3 of Graphic ES-9. In sediment and native material samples where shake tests were not performed (e.g., National Grid cores), direct visual observation of blebs, coated, or saturated NAPL, indicates that NAPL is potentially present.

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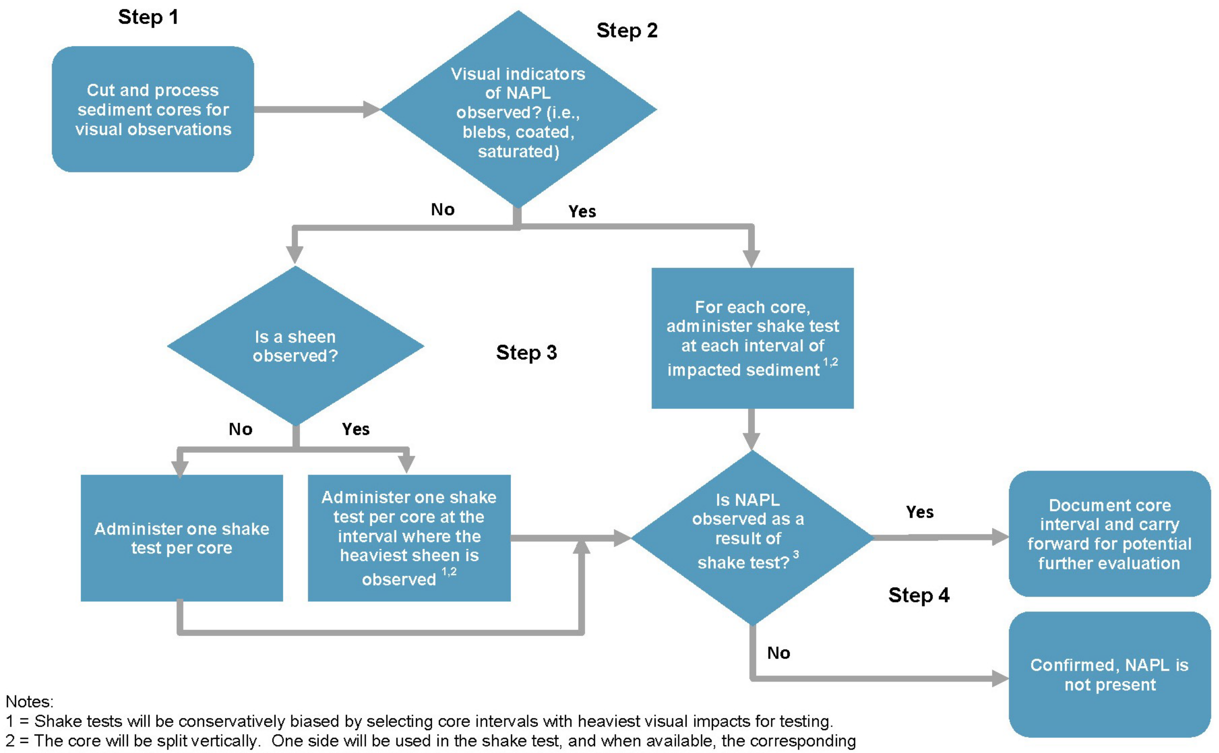
[13] A shake test consists of placing sediment and distilled water into a clean laboratory jar, which is shaken and allowed to equilibrate, to observe whether a separate phase liquid is generated.



Notes:
1 = Shake tests will be conservatively biased by selecting core intervals with heaviest visual impacts for testing.
2 = The core will be split vertically. One side will be used in the shake test, and when available, the corresponding interval of the other undisturbed portion will be retained and archived for potential further evaluation.
3 = Observation of sheen only is not considered nonaqueous phase liquid (NAPL; Anchor QEA 2014b). Observation of NAPL blebs or a NAPL layer in a shake test indicates NAPL presence.

Graphic ES-9. Flow Diagram for Field Identification of NAPL

For much of the Study Area, where NAPL was observed, NAPL observations in sediment were intermittent (i.e., located sporadically throughout an area, not clustered at a particular location) and residual (i.e., shake test blebs, bleb visual observations).[14] A relatively greater magnitude of NAPL (i.e., shake test layer results, coated and saturated visual observations) was observed in three limited areas of the Study Area, referred to as Category 2/3 Areas. Notable patterns in the data are as follows:

- **CM 0 – 2.** NAPL was not observed in surface sediment; however, sheen in surface sediment was observed intermittently in 25% (27 of 108) of surface samples collected in CM 0 – 2. In subsurface sediment, sheen and NAPL were observed more frequently at

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[14] Residual NAPL is the condition where NAPL saturation is sufficiently low that the NAPL consists of discrete blebs trapped by capillary forces, so it is immobile. This classification is specific to the ability of the NAPL to advect (i.e., flow) as a nonaqueous fluid phase. The interpretation that blebs represent residual, immobile NAPL is based on the observation that in core samples, the blebs are present as small, discrete droplets; this matches the description of residual NAPL as documented in the literature (Schwille 1988; Cohen and Mercer 1993; Pankow and Cherry 1996; API 2003; ITRC 2004; Sale et al. 2008; ITRC 2009; Kueper and Davies 2009).

various locations and depths. Where observed, NAPL was primarily in a residual state. From CM 1.6 to 1.7, shake test results for a limited number of cores indicated the presence of Category 2/3 NAPL. This area is referred to as the CM 1.7 Category 2/3 Area. With the exception of a few samples, NAPL in CM 1.7 is not present in measurably greater amounts than the surrounding areas in the CM 0 – 2 reach. NAPL mobility testing of CM 0 – 2 subsurface sediment and native material samples demonstrated that, where present, NAPL was immobile, so that NAPL will not migrate to surface sediments from underlying subsurface sediments and native material.

- **CM 2+.** NAPL was observed in surface sediment at a limited number of locations—primarily in a residual state upstream of CM 2.4. This area is referred to as the Turning Basin Category 2/3 Area. Sheen was observed in surface sediment samples at a number of surface sediment locations in this reach. In subsurface sediment, sheen and NAPL were observed more frequently than in CM 0 – 2 at various locations and depths. Quantitative NAPL mobility testing for CM 2+, including the Turning Basin Category 2/3 Area, was completed as one component of the FS Part 2 field program (data for the FS Part 2 field program are not included in the RI Report and are presented in the *Feasibility Study Nonaqueous Phase Liquid Mobility Data Evaluation Report* (FS NAPL DER; Anchor QEA 2022a).
- **Tributaries.** NAPL was not observed in surface sediment in the tributaries, except at one location, in lower English Kills. Sheen was observed in approximately half of the surface sediment samples scattered throughout the tributaries. Sheen was also observed in subsurface sediment at various depths throughout the tributaries. While NAPL was not observed in subsurface sediments in Dutch Kills and Whale Creek, it was observed in a limited number of locations in Maspeth Creek, East Branch, and the upper reach of English Kills, and more widely in the lower reach of English Kills. Category 2/3 NAPL was observed in a limited number of cores, all located in the lower portion of English Kills, between CM 2.95 and 3.2. This area is referred to as the Lower English Kills Category 2/3 Area. Quantitative NAPL mobility testing for the tributaries, including the Lower English Kills Category 2/3 Area, was performed as part of the FS Part 2 field program (data for the FS Part 2 field program are not included in the RI Report and are presented in the FS NAPL DER [Anchor QEA 2022a]).

NAPL observations in the native material were primarily limited to the areas of the Turning Basin and English Kills with footprints overlapping where NAPL was also observed in subsurface sediment. Isolated sheens in native material samples were infrequently observed in the main stem, primarily between CM 1.3 and 2.7, in lower English Kills, and at one location in Maspeth Creek.

To understand whether gas ebullition can facilitate NAPL transport from the sediment bed to surface water, qualitative studies of gas ebullition were conducted as part of the Phase 2 investigations during times of the year when gas ebullition is most active (i.e., during low tides or warmer temperatures). Observations of the location, frequency, and magnitude of bubble generation and sheen blossoms[15] at the water surface were recorded to develop an understanding of conditions where gas ebullition-facilitated NAPL transport would most likely be expected to occur. A quantitative gas ebullition pilot study was conducted in September 2017 to develop and test methodologies for the 2018 to 2019 gas ebullition field program that was conducted under Part 2 of the FS field program (data for the 2018 to 2019 field program are not included in the RI Report and are presented in the *Feasibility Study Gas Ebullition Data Evaluation Report* [FS Gas Ebullition DER; Anchor QEA 2022b]).

Sources

The current distribution of contaminants in the sediment column of the Study Area is due to historical and ongoing sources, historical dynamic fate and transport processes, and changes in contaminant loads over time. As such, the locations of impacts observed today cannot necessarily be directly linked to proximate upland sites or sources, including point sources. Historically, contaminant loads to surface sediment were much greater, as evidenced by the higher contaminant concentrations in subsurface sediment. Surface sediment concentrations have been declining over time, as a result of the deposition and mixing of these recently deposited cleaner solids with previously deposited solids. Because the constituents that describe the nature and extent of contamination are also commonly present in the urban environment of

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[15] Not all sheens on the water surface originate from ebullition. Sheen blossoms are sheens that appear with a breaking gas bubble (i.e., ebullition). There can be distinct static sheens, which float on the water surface into the observation area. Potential static sheen sources might be caused by seepage from bulkheads, floatables, outfall discharge, surface scum, vessel movements, or discharges from engine/bilge/deck runoff, as well as unknown sources.

the Study Area, these contaminants can enter the system from multiple potential sources. These sources are described in the following list, and current loads to the Study Area (by reach) are summarized for several of these sources in Table ES-1 for TPAH (17), TPCB, and Cu:

- **Point sources and overland flow.** Almost one-third of the point source TPAH (17) load (30% to 32%) enters the Study Area in CM 0 – 1 from the Con Edison – 11th Street Conduit (*Data Applicability Report* No. 110) dewatering system. The majority of the point source TPAH (17) (51%), TPCB (67%), and Cu (75%) loads enter the Study Area in the tributaries—primarily Maspeth Creek, East Branch, and English Kills—predominantly from CSOs and stormwater.
- **East River.** The East River transports solids that contain contaminant concentrations consistent with the reference areas as a load to the Study Area, due to the semidiurnal tides. The East River is the primary source of the solids that deposit on the sediment bed in CM 0 – 2 and the lower tributaries (i.e., Whale Creek and Dutch Kills); these solids, along with upstream point sources, contribute to the solids that deposit in CM 2+ and, to a lesser extent, the sediment bed in the upper tributaries (i.e., Maspeth Creek, East Branch, and English Kills). Concentrations of TPAH (17), TPCB, and Cu measured in East River surface water samples collected near the mouth of Newtown Creek are generally similar to those measured in CM 0 – 2 during dry weather, reflecting the strong influence of the river on this reach of the Study Area. Estimating the contaminant loads from the East River to the Study Area requires the use of linked hydrodynamic, sediment transport, and CFT models. This work is underway and will be included in FS-related reports.
- **Groundwater.** Groundwater discharge to the Study Area occurs at the base of the Study Area and through vertical permeable shorelines to the surface water (i.e., lateral discharge; see next bullet). The base of the Study Area is defined as the interface between sediment and native material, as well as between sediment and fill. Groundwater discharge to the base of the Study Area may provide chemical loads to subsurface sediment and surface sediment, eventually discharging to surface water. This load is a small fraction of the contaminant mass present in the subsurface sediment, meaning that the subsurface sediment chemical concentrations are from other historical legacy sources. In addition, groundwater contamination, where present, is substantially attenuated in the subsurface sediment before it reaches surface sediment. For example, the total groundwater TPAH (17) load from the base of the

Study Area to subsurface sediment in CM 2+ is estimated to be between 740 and 1,400 kilograms per year (kg/year), but the load of TPAH (17) in porewater[16] flowing from subsurface to surface sediment in this reach is approximately 100 to 200 times less (7.3 kg/year). In total, groundwater contaminant loads to the surface sediments in the Study Area are minor relative to contaminant loads from point sources.

- **Lateral groundwater discharge.** Lateral groundwater discharge through vertical permeable shorelines also may transport contaminants to the water column. However, dry weather surface water data adjacent to the five areas with the highest estimated lateral groundwater discharge rates per linear foot of shoreline indicate no observable influence from lateral groundwater discharge on surface water chemical concentrations, although definitive conclusions cannot be drawn from such comparisons. Because shallow lateral groundwater discharge inputs to Newtown Creek have not been empirically characterized, USEPA is planning a study to further characterize shallow lateral groundwater discharge along the shoreline of Newtown Creek. The stated objective of the USEPA study is to collect empirical data to achieve sufficient characterization of shallow lateral groundwater discharge to support the FS and reduce uncertainty in the current lateral groundwater discharge estimate. Chemical loads from lateral groundwater discharge will also be further evaluated with the CFT model during the FS through sensitivity analysis.
- **Other sources.** Shoreline erosion, atmospheric deposition, overwater activities, and shoreline seeps including NAPL seeps represent additional sources of contaminants to the Study Area that are evaluated as part of the RI. Analyses of data from historical studies and data collected during FS Part 1 field activities demonstrate that shoreline erosion, atmospheric deposition, and overwater activities represent minor sources of contaminants to surface water and surface sediment in the Study Area. Quantitative estimates of mass loading could not be calculated for shoreline seeps including NAPL seeps as there are no flow data for the seeps. These sources will continue to be assessed during the FS.

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[16] Shallow porewater can be impacted by tidal exchanges with surface water. Although there are no direct contaminant measures associated with such tidal exchange, multiple lines of evidence presented in the RI Report (Section 6.4.3.1.2) indicate that this process is not a primary driver of shallow porewater concentrations.

Table ES-1
Summary of Current Contaminant Loadings to Study Area

	TPAH (17)			TPCB			Cu		
	CM 0 – 2	**CM 2+**	**Tributaries**	**CM 0 – 2**	**CM 2+**	**Tributaries**	**CM 0 – 2**	**CM 2+**	**Tributaries**
Point Sources									
CSO	0.52 to 0.58	0.24 to 0.30	19 to 20	<0.01	≤0.012	0.12 to 0.27	6.0 to 6.6	3.4 to 7.6	180 to 220
Stormwater	4.5 to 5.8	2.2 to 3.2	6.6 to 8.0	0.098 to 0.17	0.033 to 0.094	0.12 to 0.24	60 to 68	28 to 37	72 to 94
Treated Groundwater	17	NA	NA	<0.01	NA	NA	2.3	NA	NA
WWTP Treated Effluent Overflow	NA	NA	0.93	NA	NA	0.050	NA	NA	33
Groundwater									
Base of Study Area	80 to 110	740 to 1,400	7.5 to 20	<0.01	<0.01	0.039 to 0.25	3.3	3.5	3.3
Other Sources	--	--	--	--	--	--	--	--	--
East River	TBD	TBD	TBD	TBD	TBD	TBD	TBD	TBD	TBD

Notes:
Units are kilograms per year.
-- = Analysis to date suggests minor contribution to Study Area based on available information (i.e., RI data and qualitative comparisons) recognizing that not all these sources could be quantified and that additional evaluations will continue during the FS.
NA = not available – Discharge type does not occur in this reach.
TBD = to be determined – Load will be calculated based on ongoing modeling analyses.

Risk and Exposure Pathways

The results of the comprehensive site-specific BHHRA and BERA provide one set of criteria to be used during selection of a Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) remedy in the FS. Human health risks were evaluated for 12 exposure scenarios. Potential risks to human health in excess of USEPA's acceptable cancer risk range and/or non-cancer hazard threshold were identified for the following exposure scenarios:

- **Study Area**
  - Cancer risks and non-cancer hazards associated with consumption by recreational anglers/crabbers of fish and crab tissue obtained from the Study Area, primarily due to tissue concentrations of PCBs in fish, and PCBs and dioxins/furans in crab
  - Non-cancer hazard for general construction worker exposure to surface sediment along the shoreline in limited areas within the Study Area, primarily due to PCBs in surface sediments in these localized areas
- **Reference areas**
  - Cancer risks and non-cancer hazards associated with consumption by recreational anglers/crabbers of fish and crab tissue obtained from reference areas, primarily due to PCBs in fish and crabs, with some contribution from dioxins/furans to non-cancer hazards in crab. The presence of human health risks in the reference areas suggests that regional exposure for migratory fish and crab species needs to be considered when evaluating risk management options for Newtown Creek.

The BERA (see Appendix I) evaluated multiple LOEs in a quantitative and qualitative weight-of-evidence approach and identified potential risks to ecological receptors as follows:

- **Study Area**
  - Surface sediment toxicity to benthic organisms in CM 0 – 2, CM 2+ and the tributaries is greater than toxicity in sediment in the four Phase 2 reference areas based on the results of the sediment toxicity testing. Toxicity at these locations may be associated with polycyclic aromatic hydrocarbons (PAHs, in particular, alkylated PAHs) in porewater, with some contribution from porewater metals (Cu, Pb, and zinc [Zn]). Based on further evaluations completed after the BERA

  was finalized, USEPA concluded that toxicity was correlated with bulk sediment concentrations of TPAH (34) and C19-C36.
  - Hazard quotient (HQ) values greater than a threshold of 1 were exceeded in CM 2+ and the tributaries for benthic fish, due to PAHs, Cu, Pb, Zn, and TPCB in porewater.
  - HQ values greater than 1 were calculated for various avian species, primarily due to dietary exposure to TPCB in CM 2+ and the tributaries.
  - HQs ranging from less than 1 to greater than 1 were calculated for bivalves, polychaetes (*Nereis virens*), blue crab (*Callinectes sapidus*), striped bass (*Morone saxatilis*), and mummichog (*Fundulus heteroclitus*), primarily due to exposure to TPCB, with some limited contribution from dioxins/furans and Cu.

- **Reference areas**
  - For the Phase 2 reference areas, potential risks were identified for blue crab, striped bass, and mummichog, primarily due to exposure to TPCB, with some limited contribution from dioxins/furans. The presence of ecological risks in the reference areas suggests that regional exposure for migratory fish and crab species need to be considered when evaluating risk management options for Newtown Creek.

It is important to note that migratory species such as striped bass, blue crab, and Atlantic menhaden (*Brevoortia tyrannus*) are exposed to contaminants both within and outside the Study Area, including exposure within and beyond the New York Harbor region. Striped bass and blue crab are the primary species consumed by recreational anglers and crabbers, whereas Atlantic menhaden, mummichog, and benthic invertebrates represent components of their food webs. TPCB in striped bass and TPCB and dioxins/furans in blue crab are the primary CERCLA hazardous substances driving potential human health risk. Moreover, both chemicals are bioaccumulative. Because TPCB is the primary risk driver in both species, TPCB is the primary focus of the evaluation of bioaccumulation and biomagnification throughout the Study Area food web. The relative contributions of Study Area and regional sources to TPCB in fish and crabs collected in Newtown Creek are an important consideration for remedial decision-making.

Key Findings and Conclusions

A key finding of this RI is that the reaches of the Study Area (CM 0 – 2, CM 2+, and each tributary) differ materially in physical characteristics, contaminant distributions, sources of solids and contaminants, relative contributions of historical versus ongoing sources, fate and transport processes, and risk. Those differences will play an important role in identifying, developing, and assessing remedial alternatives in the FS.

The nature and extent of contamination within the Study Area is affected by influences that include the following: historical and ongoing discharge, transport, and deposition of contaminants and solids from point sources; surface water and solids exchange with the East River (due to the tides); mixing (due to biological activity within the surface sediment [i.e., bioturbation]); episodic storm events that primarily affect the tributaries near the large outfalls; and marine vessel traffic, which also acts as a sediment mixing process. These influences contribute to the following notable observations of the nature and extent, sources, and fate and transport of contaminants (represented by TPAH [17], TPCB, and Cu) in the Study Area:

- **CM 0 – 2**
  - Concentrations of TPAH (17), TPCB, and Cu in surface sediment in CM 0 – 2 are generally the lowest in the Study Area and are consistent with (or approaching) reference areas, based on data from reference areas similar to Newtown Creek. Surface sediments are stable due to low current velocities. Concentrations tend to increase with depth in the subsurface sediment and are low in underlying native materials. Deposition of solids in this reach is primarily from East River tidal exchange. These solids mix with the existing shallow surface sediments that have been influenced by historical and ongoing sources typically found in urban, industrialized waterbodies. NAPL was only observed in subsurface sediments and has been demonstrated to be immobile. Minimal gas ebullition and sheen blossom formation have been observed in CM 0 – 2 during field surveys. Toxicity to benthic macroinvertebrates and risks to other ecological receptors such as fish and crab in CM 0 – 2 are similar to those in the four Phase 2 reference areas, with some exceptions. Surface water concentrations overlap with East River concentrations.

- **CM 2+**
  - Concentrations of TPAH (17), TPCB, and Cu in surface sediment are higher than in CM 0 – 2 and are above reference area concentrations. Concentrations tend to increase with depth in subsurface sediment and are generally lower in the native material. Solids deposited from CSOs and MS4s, stormwater inputs and runoff, and to some extent from East River tidal exchange, become mixed within the surface sediment layer via biological and physical processes, resulting in a blend of previously deposited and currently depositing contaminants in the surface sediment. NAPL was observed in several portions of the Turning Basin in subsurface sediment and native material, and less frequently in surface sediment. Areas of gas ebullition and sheen blossom formation were observed in the Turning Basin along the Brooklyn and Queens shorelines at water depths less than 6 meters. Toxicity to benthic macroinvertebrates and risks to other ecological receptors, such as fish and crab, are greater than in the Phase 2 reference areas. Toxicity to benthic macroinvertebrates at some locations cannot be attributed solely to porewater contaminant concentrations, but may be influenced by other stressors including low dissolved oxygen (DO), elevated porewater sulfide, and bulk sediment concentrations of complex hydrocarbon mixtures.
- **Tributaries**
  - Major CSOs present at the heads of English Kills, East Branch, Maspeth Creek, and Dutch Kills are the primary source of solids to the tributaries. Large MS4 outfalls are also located in the tributaries. Surface sediment exhibits very high total organic carbon (TOC) levels, primarily due to discharges of solids from CSO and MS4 point sources, but are also affected by influences from historical sources (both municipal and industrial). Concentrations of TPAH (17), TPCB, and Cu in surface sediment are generally higher than in CM 0 – 2 and are above reference area concentrations. Concentrations tend to increase with depth in subsurface sediment, but are lower in the native material. In Maspeth Creek, East Branch, and upper English Kills, NAPL was only observed in a few cores as residual NAPL. In a localized area within lower English Kills, NAPL was observed in coarse-grained beds in subsurface sediment and native material. Areas of gas ebullition and sheen blossom formation were observed in each of the tributaries.

> More widespread gas bubbles were observed in the tributaries, where the TOC is higher and water depths are generally shallower than in the main stem. Toxicity to benthic macroinvertebrates and risks to other ecological receptors such as fish and crab are greater than in the Phase 2 reference areas. Toxicity to benthic macroinvertebrates cannot be attributed solely to porewater contaminant concentrations, but may be influenced by other stressors including low DO, elevated porewater sulfide, and bulk sediment concentrations of complex hydrocarbon mixtures.

In summary, surface sediment contamination drives the ecological and human health risks within the Study Area. Due to the continuous deposition of sediments in the Study Area that are representative of inputs from sources consistent with an urban industrialized environment, reference area levels of CERCLA hazardous substances and other contaminants will reaccumulate in surface sediments, even after remedial action is undertaken. While the CERCLA process needs to consider the protection of human health and the environment, appropriate long-term equilibrium conditions in the Study Area must be established and factored into remedial decision-making where risk-based levels are not achievable due to the influence of ongoing external inputs of contaminants to the Study Area and in-creek processes that influence the nature and extent and fate and transport characteristics of these contaminants. There are characteristics associated with Newtown Creek, such as the physical structure, surrounding land uses, and hydrodynamic and sediment transport dynamics that are important to recognize when establishing long-term equilibrium conditions in Newtown Creek.

Specifically, conditions in Newtown Creek will continue to reflect ongoing external inputs to the Study Area that include, but are not limited to, tidal flows from the East River, point source discharges, overland stormwater flow, and other sources (such as atmospheric deposition, overwater activities, shoreline erosion, lateral groundwater discharge, and shoreline seeps [including bulkhead NAPL seeps]), which may have influences on more localized scales. Developing an understanding of long-term equilibrium conditions in Newtown Creek requires, to some degree, a comparison to conditions in waterbodies that are similar to Newtown Creek, but that are not influenced by the site-specific releases of hazardous substances and other contaminants that are the focus of the RI/FS process being conducted in the Study Area. Understanding regional conditions, in addition to understanding the contribution of ongoing external inputs to the Study Area, is necessary to understand possible future conditions of Newtown Creek.

Specifically with respect to risk to human health from consumption of fish and crab, the species consumed by people in the Study Area and used to represent human exposure in this risk assessment—namely, striped bass, white perch, and (to a lesser extent) blue crab—exhibit wide-ranging movement and are exposed to contamination present in the wider New York-New Jersey urban area. Furthermore, the food web of striped bass, white perch, and blue crab species may also be wide-ranging or largely water column-based, meaning that the base of the food web (smaller fish, phytoplankton, and zooplankton) likely accumulates contaminants from outside, as well as within, the Study Area.

PCB is the primary COPC that contributed to both cancer risk and noncancer hazard estimates in the Study Area and in the Phase 2 reference areas. These Phase 2 reference area results, along with an understanding of species migration and movement, indicate that fish and crab exposure to COPCs occurs on a regional scale, and COPCs in the species consumed by people fishing and crabbing in the Study Area likely originate in a wider regional urban area beyond just the Study Area boundaries. These regional-scale cancer risks and noncancer hazards are in the upper end of the USEPA acceptable risk range or above the USEPA acceptable risk range and exceed the hazard index threshold of 1. The cancer risks and noncancer hazards calculated for the Phase 2 reference areas provide one estimate of regional risks that could be present in the absence of Study Area-related contamination. Because of this, regional fish and crab consumption advisories currently in place that include Newtown Creek may persist in the future regardless of remedial actions completed in the Study Area.

Specifically with respect to ongoing external inputs, the East River and various point sources will continue to contribute a significant load of contaminants that are common in urban environments like Newtown Creek and the surrounding greater New York Harbor area even after any future sediment remediation. In comparison, contaminant loadings from lateral groundwater discharge and other non-point sources such as shoreline seeps and eroding shorelines are currently interpreted to be lesser contributors of these constituents to surface sediment based on available information, although they have not been directly quantified and some will be evaluated further in the FS. In addition, some upland properties may potentially contribute these constituents to the Study Area. The FS will need to evaluate the potential for the ongoing contribution of contaminants and other constituents as part of the remedy evaluation process, consistent with USEPA's first listed risk management principle, which states that significant direct and indirect ongoing sources should be identified and controlled if they have the potential to cause significant recontamination at sediment sites (Horinko 2002). As noted by USEPA guidance, "Identifying and controlling contaminant sources typically is critical to the effectiveness of any Superfund sediment cleanup" (USEPA 2005a). Influences from the East River, CSO and MS4 discharges, other point sources, overland stormwater flows, and other sources will continue over the long term into the creek. Accordingly, remedial alternatives evaluated in the FS need to assess these ongoing contributions, and any potential controls, in the context of the timing of the remedy and its long-term effectiveness. Notwithstanding the extensive dataset compiled during this RI, future investigations undertaken within the boundaries of the Study Area may indicate as yet unidentified sources that will need to be considered as remedial designs move forward.

The RI Report represents a comprehensive study that complies with the AOC entered into with USEPA for this site. The voluminous dataset supports multiple LOEs to characterize the nature and extent of contamination in the Study Area. This work also establishes a solid foundation to evaluate a combination of sustainable remedial approaches to utilize in different portions of the creek to achieve practicable risk reduction and ensure long-term success. The FS for Newtown Creek will utilize the information generated in the RI to evaluate cost-effective and sustainable remedies for Newtown Creek.

1 INTRODUCTION

The Remedial Investigation (RI) and the subsequent Feasibility Study (FS) for the Newtown Creek Study Area are being performed under an Administrative Order on Consent (AOC) entered into with the U.S. Environmental Protection Agency (USEPA). This *Remedial Investigation Report* (RI Report) represents the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) investigation findings at Newtown Creek. There are six signatories to the AOC, including the five members of the Newtown Creek Group (NCG) and the City of New York. The NCG includes Phelps Dodge Refining Corporation (PDRC); Texaco, Inc.; BP Products North America Inc.; The Brooklyn Union Gas Company d/b/a National Grid New York; and ExxonMobil Oil Corporation (ExxonMobil).

The AOC designates that the performance of the work be completed under the administration of a USEPA-approved project coordinator. This RI Report has been prepared by that USEPA-approved project coordinator, Anchor QEA, LLC, who has been retained by the NCG for the completion of this work. This RI Report has been prepared on behalf of the NCG under USEPA's oversight pursuant to the AOC.

1.1 Remedial Investigation/Feasibility Study Objectives

In accordance with CERCLA and the *National Oil and Hazardous Substances Pollution Contingency Plan* (NCP; 40 Code of Federal Regulations [CFR] § 300), the overall objective of the Remedial Investigation/Feasibility Study (RI/FS) is to collect sufficient data and information to define the nature and extent of contamination at the site, support characterization of risks to human health and the environment, and develop and evaluate effective remedial alternatives in the FS. The RI/FS will support risk management decisions and selection of a remedy following the appropriate USEPA and New York State Department of Environmental Conservation (NYSDEC) guidance documents and the principles outlined in the USEPA *Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA* (USEPA 1988) and *Contaminated Sediment Remediation Guidance for Hazardous Waste Sites* (USEPA 2005a). With this, the RI/FS will provide the basis for sound, scientifically based decisions to be made concerning the future of the Study Area. To achieve the overall objective, specific objectives were established in the *Remedial Investigation/*

Feasibility Study Work Plan (RI/FS Work Plan); those objectives are directly quoted in the following list:

1. Identify, quantify, and understand the vertical and horizontal distribution of contaminants of potential concern (COPCs) in sediment and surface water, and other constituents and stressors that may impact the ecology and quality of the Study Area sediment, water, and biota. This will include a complete characterization of all substances in the Study Area, notwithstanding whether the initial release included petroleum or any other substance. The synergistic relationships among substances will be considered to the extent necessary for such characterization.
2. Identify and quantify ongoing significant loadings of COPCs and, to the extent of the available information, sources of such loadings to the Study Area surface water, sediments, groundwater, and biota. In the case of ongoing upland sources, refer future investigation of sources to the appropriate regulating agency (i.e., the USEPA, the NYSDEC, or the NYC Department of Environmental Protection [NYCDEP]). For more details on evaluation of upland sources, see Section 3.2.4 [RI/FS Work Plan]. As stated in USEPA *Contaminated Sediment Remediation Guidance for Hazardous Waste Sites* (USEPA 2005a), sources of contaminants to sediments must be controlled early and if recontamination is likely to occur, then sources should be controlled prior to establishing end points and prior to the implementation of sediment remedies. Therefore, it is important to identify and control significant sources of contaminants to the Study Area, prior to implementing an effective remedy.
3. Understand the key geomorphological, chemical, and biological processes affecting the stability of sediments and the fate, transport, and bioavailability of COPCs.
4. Identify complete and reasonably potentially complete (considering the urban nature of the Study Area and the impact of future contaminant loadings on the ecology and quality of the Study Area) exposure pathways and identify potential current and future human

health and ecological risks posed by the COPCs present in the Study Area.

5. Identify and evaluate potential remedial actions that provide meaningful risk reduction and provide the highest, best possible use of the Study Area, and that also consider the urban nature of the Study Area and the impact of future contaminant loadings on the ecology and quality of the Study Area. (AECOM 2011)

As described in Section 1.4 (Phased Investigation Approach) of the RI/FS Work Plan (AECOM 2011), the approach for completing the RI/FS includes several phases of field investigations, along with associated evaluations and reporting (consistent with USEPA guidance). For the purposes of this RI Report, the Phase 1 and Phase 2 RI portions of the program are referred to as Phase 1 and Phase 2 throughout the document and associated appendices. In addition, Part 1 of the FS field program is included in this RI Report and referred to throughout as FS Part 1 or Part 1 of the FS. The results of the Phase 1 work were reported in the *Phase 1 Remedial Investigation Interim Data Report* (Phase 1 IDR; Anchor QEA 2012a) and *Phase 1 Remedial Investigation Field Program Data Summary Report – Submittal No. 3* (Phase 1 DSR Submittal No. 3; Anchor QEA 2013a). The *Phase 2 Remedial Investigation Work Plan – Volume 1* (Phase 2 RI Work Plan Volume 1; Anchor QEA 2014a) and *Phase 2 Remedial Investigation Work Plan – Volume 2* (Phase 2 RI Work Plan Volume 2; Anchor QEA 2014b) considered Phase 1 data to inform field data collection needs for Phase 2 and also updated the conceptual site model (CSM) contained in the RI/FS Work Plan (AECOM 2011). The results of Part 1 of the FS are included in the *Feasibility Study Field Program Data Summary Report* (FS DSR Part 1; see Appendix Bii), consistent with the approach developed in the *Feasibility Study Field Program Work Plan* (FS FP Work Plan; Anchor QEA 2017a).

All five of the following study objectives have been met in this RI or will be met in the FS:

- Objective 1: Characterize contaminant distribution—Section 4 of this RI Report
- Objective 2: Characterize and quantify ongoing significant loads—Section 5 of this RI Report, the Point Sources Evaluation (see Appendix E), and the Groundwater Evaluation (see Appendix F)

- Objective 3: Understand stability, fate, and transport—Section 6 of this RI Report—and the numerical modeling studies documented in Appendix G and being performed as part of the FS
- Objective 4: Characterize risks—Section 7 of this RI Report, the *Baseline Ecological Risk Assessment* (BERA; see Appendix I), and the *Baseline Human Health Risk Assessment* (BHHRA; see Appendix H)
- Objective 5: Identify remedial actions—to be addressed in the FS

1.2 Study Area

The Study Area is defined in the AOC as Newtown Creek and its tributaries (Dutch Kills, Maspeth Creek, Whale Creek, East Branch, and English Kills). It is approximately 3.8 miles long and extends to the ordinary high water (OHW) mark (see Figure 1-1).[17,18]
Newtown Creek and its five tributaries form part of the border between the boroughs of Brooklyn and Queens, New York City (NYC), New York.

1.3 Site Background

Historical records indicate that the Newtown Creek area was inhabited by the Mispat tribe in the early 1600s. In 1613, the Dutch, as the New Netherland Company, established fur trading posts in New York. By 1623, the New Netherland Company began selling tracts of land around Newtown Creek. The earliest land grants included Greenpoint, which was purchased in 1645. On the Queens side, modern day Hunter's Point, located at the mouth of

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[17] The Newtown Creek Superfund Site Study Area is described in the AOC as encompassing the body of water known as Newtown Creek, situated at the border of the boroughs of Brooklyn (Kings County) and Queens (Queens County) in the City of New York and the State of New York, roughly centered at the geographic coordinates of 40° 42' 54.69" north latitude (40.715192°) and 73° 55' 50.74" west longitude (-73.930762°), having an approximate 3.8-mile reach, including Newtown Creek proper and its five branches (or tributaries) known respectively as Dutch Kills, Maspeth Creek, Whale Creek, East Branch, and English Kills, as well as the sediments below the water and the water column above the sediments, up to and including the landward edge of the shoreline, and including also any bulkheads or riprap containing the waterbody, except where no bulkhead or riprap exists, then the Study Area shall extend to the OHW mark, as defined in 33 CFR § 328(e) and the areal extent of the contamination from such area, but not including upland areas beyond the landward edge of the shoreline (notwithstanding that such upland areas may subsequently be identified as sources of contamination to the waterbody and its sediments or that such upland areas may be included within the scope of the Newtown Creek Superfund Site as listed pursuant to Section 105(a)(8) of CERCLA).

[18] The term "creek" is used interchangeably with "Study Area" throughout this RI Report.

Newtown Creek, was sold in 1652. By the mid-1600s, the area was primarily occupied by farmsteads and small agrarian villages, and the waterway was used for local travel. Early industry at this time included brick, potash, timber, and shipbuilding. In 1731, it was reported that the principal trade items were furs, whalebone, pitch, and tar. To continue the supply of fresh produce to Brooklyn and Manhattan, a public dock was established in Maspeth called "English Kills Landing" in 1769. Two stores were located here by the 1790s. By the 1800s, farms and plantations lined the shores of the creek (NYSDOT and FHWA 2005; Goodwin and Associates 2012).

Industrial activities in the surrounding uplands and use of the creek for shipping and navigational purposes began in approximately 1850 and increased steadily after the American Civil War (Goodwin and Associates 2012). By 1870, the creek was already heavily industrialized, and additional manufacturing operations followed into the close of the nineteenth century and the beginning of the twentieth century. These included petrochemical plants; animal rendering operations, such as fertilizer and glue factories; copper smelting; sugar refining; pencil manufacturing; manufactured gas production; sawmills; and lumber and coal yards. The area hosted many commercial vessels, including large boats bringing in raw materials and taking out manufactured products, chemicals, and metals.

Direct discharge to Newtown Creek was the primary method for disposal of stormwater, sewage, and industrial wastewater prior to the early twentieth century. Historical industrial discharges to Newtown Creek included waste liquor from grease vats and digesters at fertilizer and fat rendering facilities; oily wash water generated during the kerosene treating process at refineries; condensate from exhaust hoods at varnish and printing works; effluent from air-scrubbing systems; and cooling water, plating wastes, and plant clean-up water from the manufacture of electrical wiring devices (BPL 1890; Hazen and Sawyer 1960; Hurley 1994; NYSL 1900; Baker and Kent 1887). Early municipal sewer systems also discharged directly to the creek. In the late 1800s and the first half of the twentieth century, direct discharges from sewers operated by NYC regularly added raw sewage and other pollutants into the creek. By 1910, intercepting sewers (i.e., interceptors) had been constructed in some areas near the creek to convey flows to the East River where flows were discharged without treatment. However, stormwater, sewage, and industrial wastewater continued to discharge to the creek via relief outfalls in areas without interceptors and when the capacity of the interceptor pipes was exceeded (Metropolitan Sewerage Commission of New York 1910; Board of Water Commissioners 1867; War Department 1891, 1915, 1936a; USEPA 2011). Since 1967, when the Newtown Creek wastewater treatment plant (WWTP) began operating, stormwater (from areas served by combined sewer systems), sewage, and industrial wastewater flows have been conveyed to either the Bowery Bay WWTP or Newtown Creek WWTP for treatment prior to discharge outside the Study Area. However, in some portions of the Study Area, direct discharges of stormwater from private sites and municipal separate storm sewer systems (MS4s), as well as combined sewer overflows (CSOs), continued and are ongoing today (see Section 3.2.8 for more details on discharges to Newtown Creek).

In addition to historical industrial and municipal discharges, Newtown Creek and its tributaries have also been impacted by spills. NYSDEC maintains a database of petroleum spills dating back to 1978, and since that time, there have been at least 30 spills documented as direct spills to Newtown Creek and its tributaries (see Section 3.2.11 for more details on historical spills). There are no records of pre-1978 petroleum spills or records of spills of any other type of hazardous materials.

Significant changes have occurred in the use of Newtown Creek and the surrounding uplands since the early 1800s. These changes have resulted in filling and major reworking of the banks and channel for drainage, industrial and municipal discharges, and navigation. Historically, freshwater flow to the creek included two components, tributary flow and groundwater flow. With centuries of industrial, commercial, and residential development, Newtown Creek is fully urbanized, with no remaining natural marshlands or freshwater streams. Instead, freshwater flows are dominated by CSO and other point source discharges (i.e., WWTP treated effluent overflows,[19] stormwater [including overland flow], and treated groundwater effluent from remediation and dewatering systems[20]) and groundwater.

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[19] With regard to "treated effluent overflows," the Newtown Creek WWTP began operating in 1967 and employed a modified aeration treatment process. Upgrades to achieve secondary treatment were completed in 2011. Although the treated effluent from the Newtown Creek WWTP is normally discharged outside the Study Area to the East River, under certain high-flow conditions (which are described in Section 3.2.8), treated effluent from the Newtown Creek WWTP is discharged inside the Study Area via the high-relief outfall (NCB-002) to Whale Creek (Cunetta and Feuer 1968; Greeley and Hansen et al. 2010; NYCDEP 2011b, 2017).

[20] With regard to "treated groundwater effluent," the degree of treatment is variable and documented in each individual State Pollutant Discharge Elimination System (SPDES) permit.

Much of the past manufacturing has ceased and new commercial and industrial operations have begun; many upland sites are also in different stages of investigation and remediation under the various NYSDEC cleanup programs (e.g., voluntary cleanup program, state superfund program). USEPA conducted an Expanded Site Investigation of Newtown Creek (excluding tributaries) in 2009, as part of the Superfund Hazard Ranking System scoring process. USEPA concluded that the variety and distribution of chemicals indicated origins from multiple sources over time. Potential sources identified included current and historical industrial and municipal discharges from shoreline and watershed upland locations; historical dredge and fill practices; historical discharge practices of industrial wastes and sanitary sewage; overwater loading and unloading; and ongoing releases from CSOs and navigational and maritime activities. The Newtown Creek Superfund Site was listed on the National Priorities List on September 29, 2010 (USEPA 2011).

Today the predominant land use around Newtown Creek and its tributaries remains industrial, with pockets of mixed use, commercial, and residential developments (NYCDEP 2011a). Current uses near the creek include the following: warehouse and distribution facilities; vehicle storage and maintenance; electrical distribution; plastics and foil manufacturing; waste transfer yards and recycling facilities; road service support facilities; construction materials storage; facilities that store electrical equipment; scrap metal processing facilities; lumberyards; ready-mix concrete plants; bulk fuel distribution terminals; railroads (e.g., tracks, yards); utilities; and municipal wastewater treatment (see Appendix J; Anchor QEA 2014a, 2014b).

Future primary land uses around Newtown Creek are expected to remain largely industrial, as the site is designated one of six NYC Significant Maritime and Industrial Areas (SMIAs; NCBOA 2012). The Newtown Creek SMIA is the largest of the six SMIAs, encompassing approximately 780 acres (NYC 2011). As recently as 2015, the City of New York's mayor, the Honorable Bill de Blasio, announced that the City's industrial properties were to be preserved and a plan to grow the City's industrial and manufacturing jobs had been prepared (CBS News 2015). The City of New York has designated some areas surrounding Newtown Creek for housing development, including Greenpoint-Williamsburg in Brooklyn and Hunter's Point, located at the mouth of the creek in Queens. Hunter's Point was recently rezoned and is in development for residential use (USEPA 2011; NYCEDC 2016a).

1.4 Report Organization

The remaining sections of this report include the following.

Section 2 – Program Summary

The Program Summary provides a review of the Newtown Creek RI field data collection program, which was conducted in two phases, and Part 1 of the FS data collection program. Phase 1 of the RI was conducted between October 2011 and September 2013 and was intended to broadly characterize key chemical and physical features of the Study Area. Phase 2 of the RI was conducted between May 2014 and December 2015 to fill data gaps and collect additional data needed to support the risk assessment, modeling, point sources, nonaqueous phase liquid (NAPL), gas ebullition, and groundwater evaluations. Part 1 of the FS was conducted between May 2017 and April 2018 to collect additional data to support the development and evaluation of remedial alternatives. The timing of the completion of a subset of the FS data collection program (Part 1 of the FS) has made it possible to include these data in this RI Report to further refine the CSM.

The phased RI and FS activities are also described in the RI/FS Work Plan (AECOM 2011) and were conducted consistent with the following documents developed as part of the RI/FS process:

- Phase 1 *Field Sampling and Analysis Plan* (FSAP; Anchor QEA 2011a)
- Phase 1 *Quality Assurance Project Plan* (QAPP; Anchor QEA 2011b)
- Phase 1 FSAP Addenda Nos. 1 through 7 (Anchor QEA 2012b, 2012c, 2012d, 2012e, 2013b, 2013c, 2013d)
- *Phase 1 Remedial Investigation Work Plan Addendum* (WPA) and associated memoranda (*Phase 1 Remedial Investigation Work Plan Addendum: Reference Area Memorandum* and *Phase 1 Remedial Investigation Work Plan Addendum: Reference Area Memorandum No. 2*; Anchor QEA 2012f, 2012g, 2012h)
- Phase 1 RI/FS Field Program – QAPP/FSAP Deviation Memoranda Nos. 1 through 5 (Anchor QEA 2012i, 2012j, 2012k, 2012l, 2012m)
- Phase 2 RI Work Plan Volume 1 (Anchor QEA 2014a)
- Phase 2 RI Work Plan Volume 2 (Anchor QEA 2014b)
- *Final Groundwater Investigation Work Plan* (USEPA 2014a)

- *Phase 2 Field Sampling and Analysis Plan – Volume 1* (Phase 2 FSAP Volume 1; Anchor QEA 2014c)
- *Phase 2 Field Sampling and Analysis Plan – Volume 2* (Phase 2 FSAP Volume 2; Anchor QEA 2014d)
- *Phase 2 Quality Assurance Project Plan* (Phase 2 QAPP; Anchor QEA 2014e)
- Phase 2 FSAP Volume 1 – Addendum No. 1 (Anchor QEA 2014f)
- Phase 2 FSAP Volume 2 – Addenda Nos. 1, 2, 3, and 4 (Anchor QEA 2014g, 2015a, 2015b, 2016a)
- Phase 2 Field Program – QAPP/FSAP Deviation Memoranda Nos. 1 through 14 (Anchor QEA 2014h, 2014i, 2014j, 2014k, 2014l, 2015c, 2015d, 2015e, 2015f, 2015g, 2015h, 2015i, 2015j, 2016b)
- FS FP Work Plan (Anchor QEA 2017a)
- *Feasibility Study Quality Assurance Project Plan* (FS QAPP; Anchor QEA 2017b)
- *Feasibility Study Field Sampling and Analysis Plan* (FS FSAP; Anchor QEA 2017c)

Section 3 – Environmental Setting

This section describes the physical environment of the Study Area, including human influences and land use. Geology, hydrogeology, sediment bed characteristics, bathymetry, hydrodynamics, sediment transport, water quality, and habitat are discussed. Section 3 also reviews the historical and current creek and upland uses, creek configuration, dredging, navigational history, and municipal and non-municipal conveyance systems and discharges through time.

Section 4 – Nature and Extent of Contamination

Section 4 discusses the distribution of contaminants in Study Area surface and subsurface sediment, surface water, porewater, groundwater, tissue, and ambient air. Reference area data are described as well. While many contaminants are present in the Study Area, this section presents and describes data for contaminants that were found to contribute to ecological and/or human health risk. These contaminants are as follows:

- Total polycyclic aromatic hydrocarbon (17) (TPAH [17])
- Total polycyclic aromatic hydrocarbon (34) (TPAH [34])
- C19-C36 aliphatic hydrocarbons (C19-C36; a class of compounds that is representative of hydrocarbons having between 19 and 36 carbon atoms that is also a component of

C9-C40 total petroleum hydrocarbons [TPH], which also includes the C10-C28 diesel range organics [DRO])

- Total polychlorinated biphenyl (TPCB)
- 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD)[21]
- Copper (Cu)
- Lead (Pb)
- Dieldrin

The RI uses this list of contaminants to help describe, represent, and understand the nature and extent of contamination in the Study Area and reference areas. However, not all of these contaminants contribute to ecological and human health risk to the same degree. The primary drivers of ecological and human health risk are hydrocarbons and polychlorinated biphenyls (PCBs), with Cu and the other constituents in the list also contributing to risk. Additionally, within a given class (such as hydrocarbons, bioaccumulative organics, and metals) these contaminants exhibit correlation and collocation spatially (to varying degrees and over varying spatial scales) as well as similar fate and transport properties. As such, the RI focuses on TPAH (17), TPCB, and Cu as a representative subset of contaminants for subsequent evaluations of sources, fate and transport, and the overall CSM.

Section 5 – Sources

This section discusses the potentially significant ongoing sources to the creek, including point sources, groundwater, the East River, bank erosion, atmospheric deposition, overwater activities, and contaminant seeps. Chemical load calculations for TPAH (17), TPCB, and Cu are also presented in this section.

Section 6 – Fate and Transport

Section 6 provides a discussion of the physical and chemical processes that govern the movement of contaminants in the Study Area. Topics include hydrodynamics, sediment transport, partitioning, surface sediment/water exchange processes (e.g., deposition and

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[21] The risk assessments identified total dioxin/furan TEQ as a driver of risk associated with human consumptions of fish and crabs. Given that 2,3,7,8-TCDD is a major contributor to total dioxin/furan TEQ, 2,3,7,8-TCDD is the congener selected for the RI nature and extent characterization.

resuspension), subsurface sediment processes, and bioaccumulation. A mass inventory and load analysis of TPAH (17), TPCB, and Cu is presented in this section as well.

Section 7 – Risk Assessment Summary

This section provides a summary of key findings from the final BHHRA and BERA, which are Appendices H and I, respectively.

Section 8 – Conceptual Site Model

Section 8 integrates the results of all of the previous sections into a description of the pathways and processes by which contaminants (focusing on the subset of TPAH [17], TPCB, and Cu) move throughout the various components and media of the entire physical, chemical, and biological system. The CSM expands upon previous CSMs contained in the RI/FS Work Plan (AECOM 2011) and the Phase 2 RI Work Plan Volumes 1 and 2 (Anchor QEA 2014a, 2014b), and it is the foundation for decision-making in the FS. The CSM addresses historical and current activities, potential sources of contamination, and how they relate to sediment, water, and biota in the system. As per Section 2.0 of the RI/FS Work Plan (AECOM 2011), the CSM will eventually be used as a tool to help select appropriate remedies for the Study Area.

Section 9 – Conclusions

The conclusions of the RI (including Part 1 of the FS) are presented in this section.

Section 10 – References

The references for the RI (including Part 1 of the FS) are contained in this section.

The following 10 appendices are included with this RI Report:

- **Appendix A – Supplemental RI Report Information**. This appendix contains sets of tables and figures for an expanded list of chemicals, presenting surface sediment, subsurface sediment, native material, sediment trap, surface water, porewater, and tissue data from Sections 4, 5, and 6, indexed by RI Report section.
- **Appendix B – Phase 1 IDR and All DSRs**. The Phase 1 IDR and the Data Summary Reports (DSRs) for Phase 1 and Phase 2 include RI program descriptions and are included as Appendix Bi. The FS DSR Part 1 (see Appendix Bii) includes program

descriptions for the FS-related field activities (Part 1 of the FS) included in this RI Report. The DSRs contain electronic versions (electronic data deliverables [EDDs]; supplied on DVD) of the chemical database for the RI and Part 1 of the FS. Appendix Biii includes a supplemental data usability assessment for data collected during Phase 1 of the RI (Anchor QEA 2013a).

- **Appendix C – NAPL Evaluation**. To determine the nature and extent of NAPL in the Study Area, NAPL data collected during the RI and Part 1 of the FS, relevant data from investigations conducted by others in the Study Area, physical characteristics of the Study Area, and the locations of potential point source discharges and industrial activities on upland sites adjacent to the Study Area were compiled and evaluated. The detailed evaluations of the nature and extent of NAPL in areas with residual and more substantial NAPL observations are provided in Appendix C.
- **Appendix D – Gas Ebullition Evaluation**. This appendix discusses the gas ebullition field surveys conducted in 2015 and 2016 and the pilot study conducted in 2017 to assess the portions of the Study Area where gas ebullition-facilitated NAPL/ contaminant transport may be occurring, and under what conditions it is most likely to occur. A summary of the approach, results, and implications for the FS is provided in this appendix.
- **Appendix E – Point Sources Evaluation**. This appendix presents the results of the point sources sampling and analysis program. Results presented in this appendix include the relative magnitude, spatial variations, and temporal variations in point sources (i.e., CSOs, WWTP treated effluent overflow, stormwater [including overland flow], and treated groundwater effluent from remediation and dewatering systems) to the Study Area, as well as the approaches for calculating loads and estimates of point source loads to the Study Area. Additional in-creek measurements were considered as further lines of evidence (LOEs) in the interpretation of the data from the point sources sampling and the resulting load estimates.
- **Appendix F – Groundwater Evaluation**. This appendix discusses groundwater interactions with Newtown Creek and its tributaries. Groundwater discharge is evaluated in a three-tier process, with an increasing level of detail and site specificity in successive tiers. Groundwater flow rates are combined with groundwater quality data to estimate mass loads of TPAH (17), TPCB, and Cu to the sediment from groundwater. The results of the groundwater evaluation for flow rates are used as

inputs to the hydrodynamic model, and the results will also be used in the contaminant fate and transport (CFT) model being developed as part of the FS to assess mass loading to surface sediment, surface porewater, and surface water.

- **Appendix G – Final Modeling Results Memorandum (FMRM)**. This appendix presents the development, calibration, and validation of the hydrodynamic and sediment transport models that will be applied to the Study Area. A propeller wash (propwash) resuspension submodel has been developed and incorporated into the sediment transport model. A discussion of the development, calibration, and application of the geographically neutral (geo-neutral) point sources model (2013 and 2015 versions) is also included in the appendix. The hydrodynamic, sediment transport, and the geo-neutral point source models (and submodels) are included as part of the RI Report.
- **Appendix H – Baseline Human Health Risk Assessment**. The BHHRA presents risk estimates for a number of exposure scenarios where people (e.g., recreational boaters/swimmers, anglers/crabbers, and occupational landside/dockside workers) could potentially be exposed to COPCs found in surface sediment, surface water, fish and crab tissue, and air in the Study Area.
- **Appendix I – Baseline Ecological Risk Assessment**. The BERA presents risk estimates for aquatic life (e.g., benthic macroinvertebrates, bivalves, fish, and crabs) and wildlife (e.g., aquatic-dependent birds) that could potentially be directly exposed to contaminants of potential ecological concern (COPECs) in surface sediment and surface water in the Study Area and indirectly exposed through their diet.
- **Appendix J**. Contains a summary list of the draft *Data Applicability Report* (DAR) and addenda previously submitted for inclusion in the administrative record, as well as the draft DAR; Appendices A, B, and C to the draft DAR; and the 2018 upland site summaries. The draft DAR provides a summation of the historical data collection and review process undertaken to date, consistent with the RI/FS *Data Collection Plan* (Anchor QEA 2011c) and the RI/FS Work Plan (AECOM 2011). The draft DAR reviews the collection and evaluation of historical documentation and data pertaining to the Study Area itself, as well as the collection and evaluation of pre-existing documents and data from upland areas adjacent to the Study Area. The upland area data collection activity was intended to help identify potential significant sources of contamination to the Study Area.

2 PROGRAM SUMMARY

Newtown Creek and its tributaries have been studied extensively as part of the RI program, Part 1 of the FS, and various in-creek NYSDEC-directed investigations. These studies, along with other regional investigations included in the RI, are summarized here and detailed in Table 2-1:

- RI field program studies
    - Phase 1, which provided an initial characterization of the Study Area
    - Phase 2, which filled data gaps identified after Phase 1 (modeling, point sources, NAPL, groundwater, and gas ebullition), and collected information to support the risk assessments
- Part 1 of the FS field program studies
    - Portions of the FS field program studies including groundwater seepage, shoreline sediment, gas ebullition pilot study, NAPL distribution refinement, and NAPL mobility (creek mile [CM] 0 – 2)[22]
- Other field program studies
    - National Grid in-creek sediment data related to an upland investigation (GEI 2009, 2010)
    - Various investigative studies within the region, including sediment, soil, groundwater, tissue, air, and surface water

These studies are the focus of this RI Report. The timing of the completion of a subset of the FS data collection program (Part 1 of the FS) has made it possible to include these data in this RI Report in order to further refine the CSM. Part 2 of the FS studies, which include geotechnical, gas ebullition, biota tissue, and NAPL mobility testing (CM 2+ and tributaries) are presented in the *Feasibility Study Geotechnical Data Evaluation Report* (FS Geotechnical DER; Anchor QEA 2020b), FS NAPL DER (Anchor QEA 2022a), and FS Gas Ebullition DER (Anchor QEA 2022b).

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[22] Creek miles are shown in Figure 1-1; further discussion of creek miles and reaches is provided in Section 4.1.

The design of the RI sampling program was based (in part) on an evaluation of reports, published literature, and media pertaining to the Newtown Creek watershed. These studies are summarized in the draft DAR (see Appendix J).

In total, more than 4,350 samples and 1,000,000 individual analytical measurements from more than 900 locations have been collected by the NCG within the Study Area and the nearby region during the RI and Part 1 of the FS field program. These data, along with select data from the additional investigative studies conducted in the region (see Table 2-1), provide a comprehensive dataset with which to examine and understand the physical, chemical, and biological characteristics of Newtown Creek. The objectives and a summary of the sampling conducted as part of these studies are described in the following sections and detailed in Appendix B. Appendix B presents details of the field activities, including data collection methods and procedures, sampling results, and data usability assessments for the phases of field work included in this RI Report.

2.1 RI and Part 1 of the FS Field Program Studies

As described previously, the Newtown Creek RI field data collection program was conducted in two phases (Phase 1 and Phase 2) and Part 1 of the FS field program in one phase, with all studies following methods and procedures described in USEPA-approved work plans and conducted directly under USEPA oversight. Phase 1 sampling was conducted between October 2011 and September 2013 and was intended to broadly characterize chemical and physical features of the Study Area. Phase 2 sampling was conducted between May 2014 and December 2015 to fill data gaps, as well as collect additional data needed to support the risk assessments, modeling, point sources, NAPL, and groundwater evaluations. Part 1 of the FS field program was conducted between May 2017 and April 2018 to collect data to support the development and evaluation of remedial alternatives. Part 2 of the FS field program studies, which include geotechnical, gas ebullition, biota tissue, and NAPL mobility testing (CM 2+ and tributaries), are presented in the FS Geotechnical DER (Anchor QEA 2020b), FS NAPL DER (Anchor QEA 2022a), and FS Gas Ebullition DER (Anchor QEA 2022b). For the purposes of this RI Report, the Phase 1 and Phase 2 RI portions are referred to as Phase 1 and Phase 2 throughout the document and associated appendices, whereas the FS field program portions are referred to as FS Part 1 or Part 1 of the FS. A summary of the samples included in the RI, including the data quality objectives (DQOs) for each program and whether those DQOs were met, is provided in Tables 2-2a through 2-2f. Sample locations are presented in Figure 2-1.

Phase 1 field work included multiple physical and ecological surveys, as well as surface water, sediment, and air sampling in the Study Area and Phase 1 reference areas. Phase 1 reference areas refer to 14 waterbodies within the New York Harbor area that were under consideration for chemical and biota sampling as part of Phase 2 to develop a reference envelope. Based on the results of the Phase 1 field work, and with input from the NCG, four of the Phase 1 reference areas were selected by USEPA for use in Phase 2 sampling to support the BERA and BHHRA (see Section 2.1.2). Selection was based on similar physical and habitat characteristics to Newtown Creek, as well as an evaluation of sediment chemistry. The Phase 1 field activities were identified in the RI/FS Work Plan (AECOM 2011) and further described in the FSAP (Anchor QEA 2011a), QAPP (Anchor QEA 2011b), and associated FSAP addenda (Anchor QEA 2012b, 2012c, 2012d, 2012e, 2013b, 2013c, 2013d). The WPA (Anchor QEA 2012f) included further sampling to assist in addressing select data gaps identified in the historical data review (Anchor QEA 2012n). The results of the Phase 1 field activities and analysis of data usability were presented in the Phase 1 IDR (Anchor QEA 2012a) and Phase 1 DSR Submittal No. 3 (Anchor QEA 2013a), which includes data presented in DSR Submittal Nos. 1 and 2. The Phase 1 IDR and Phase 1 DSR Submittal No. 3 are included in Appendix Bi (see Attachment Bi-A).

The Phase 2 field activities included multiple physical and ecological surveys, as well as surface water, porewater, groundwater, point sources, sediment, and tissue sampling in the Study Area and Phase 2 reference areas. Four Phase 2 reference areas were selected—Westchester Creek, Head of Bay, Spring Creek, and Gerritsen Creek—to represent the range of physical, chemical, and habitat characteristics of Newtown Creek. Westchester Creek is most similar to Newtown Creek because of its surrounding commercial/industrial land use and the presence of CSOs. Gerritsen Creek is least similar to Newtown Creek, because its surrounding land use is undeveloped, consisting of a golf course, marine park, and the Gerritsen Creek Ecosystem Restoration Project. Furthermore, there are no CSOs present in Gerritsen Creek. Head of Bay and Spring Creek represent reference areas with characteristics between these two extremes. Land use surrounding Head of Bay consists of urban residential and commercial/industrial development, including the John F. Kennedy International Airport, but with no CSOs. Land use surrounding Spring Creek is a combination of developed (but non-industrial) and undeveloped areas, with a CSO at its upstream end.

The Phase 2 studies were described in the Phase 2 RI Work Plan Volumes 1 and 2 (Anchor QEA 2014a, 2014b), USEPA *Final Groundwater Investigation Work Plan* (USEPA 2014a), Phase 2 FSAP Volumes 1 and 2 (Anchor QEA 2014c, 2014d) and associated addenda (Anchor QEA 2014f, 2014g, 2015a, 2015b, 2016a), and Phase 2 QAPP (Anchor QEA 2014e). The results and data usability of the Phase 2 field activities are presented in the *Phase 2 Remedial Investigation Field Program Data Summary Report* (Phase 2 DSR), which is included as Appendix Bi to this RI Report. The DSRs for Phases 1 and 2 (see Appendix Bi) contain EDDs of the chemical database for Phase 1 and Phase 2.

Part 1 of the FS included groundwater seepage measurements, shoreline sediment and opportunistic seep sampling, a gas ebullition pilot study, and sediment sampling to refine NAPL distribution in the Study Area, as well as to determine NAPL mobility in CM 0 – 2. Part 1 of the FS was identified in the FS FP Work Plan (Anchor QEA 2017a) and further described in the FS QAPP and FS FSAP (Anchor QEA 2017b, 2017c). The results of the FS and analysis of data usability are presented in the FS DSR Part 1, which is included as Appendix Bii to this RI Report.

The RI and Part 1 of the FS are summarized in Sections 2.1.1 through 2.1.7 by media and are briefly described in Tables 2-2a through 2-2f. Acceptance criteria for RI and Part 1 of the FS data are documented in the approved project QAPPs and FSAPs. Data completeness goals for both phases of the RI and Part 1 of the FS were met with greater than 99% analytical completeness for each phase of the project, as summarized in Tables 2-3a through 2-3b. Deviations from the USEPA-approved work plans are summarized in Table 2-4 by field program and tabulated in Appendices Bi and Bii.

2.1.1 Physical Surveys

The physical surveys conducted in the Study Area include hydrographic surveys, shoreline surveys, Light Detection and Ranging (LiDAR), high-resolution aerial imagery, and thermal infrared (TIR) imagery. These surveys are briefly described in Sections 2.1.1.1 through 2.1.1.4 and are summarized in Table 2-2a.

2.1.1.1 Hydrographic Surveys

Hydrographic surveys were performed to support several aspects of the RI, including field sampling activities, evaluation of sediment depositional history, CSM development, and development of mathematical models of the Study Area. Hydrographic survey information was collected throughout the Study Area in the fall of 2011 and included a bathymetric survey, side-scan sonar survey, and magnetometer survey. Additional bathymetric survey work was performed in December 2012, after Hurricane Sandy,[23] to evaluate the potential impacts of a significant weather event on sediment elevations in the Study Area.

2.1.1.2 Shoreline Surveys

Shoreline surveys were conducted in 2011 (dry weather) and in 2012 and 2013 (wet weather) to identify potential sources of contaminants to the Study Area and potential source pathways, as well as to provide a preliminary survey of the physical and ecological features of the Study Area. Opportunistic sampling (i.e., sampling completed during dry weather if sufficient flow volume was present and the sampling point was safely accessible) was conducted for point source discharges and overland flow that were observed during the 2011 survey.

2.1.1.3 Light Detection and Ranging and High-Resolution Aerial Imagery

LiDAR and orthophotographic surveys were coincidently conducted by helicopter in 2012, to provide topographical information and high-resolution imagery for areas immediately adjacent to the Study Area. The LiDAR and bathymetry data were integrated together to create a seamless digital elevation model of the Study Area. The orthophotographic imagery was collected for use in conducting a detailed analysis of the shoreline areas.

2.1.1.4 Thermal Infrared Imagery

An aerial TIR imagery survey was conducted in 2012. The TIR imagery survey was performed in the summer during a period of low water and high surface water temperatures

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[23] Hurricane Sandy caused extensive flooding throughout the region and had the potential to affect sediment elevations by erosion and/or deposition. Comparison of bathymetric output pre- and post-storm showed no significant change that could be detected, beyond the resolution of the data collected.

to identify temperature variations that might suggest potential groundwater discharge areas to the Study Area. This information was used to identify Phase 2 sampling locations as part of a further assessment of groundwater.

2.1.2 Phase 1 and Phase 2 Reference Areas

Potential reference areas were investigated consistent with the *Contaminated Sediment Remediation Guidance for Hazardous Waste Sites* (USEPA 2005a) to serve as points of comparison with the Study Area. Reference areas are an important component of the RI and associated risk assessments, especially given the urban setting within which Newtown Creek is situated. During development of the Phase 2 RI Work Plan (Anchor QEA 2014a, 2014b), a list of waterbodies within the general region (but located outside) of the Study Area were compiled for a desktop assessment, as a first step to select reference areas for chemical and biota sampling in Phase 2. Based on general features and attributes of waterbodies with similar physical and habitat characteristics to Newtown Creek, a reconnaissance study (including surface sediment sampling and water quality profiling) was completed at 14 Phase 1 reference areas, as shown in Figures 2-2 through 2-16. Based on the results of that reconnaissance, and input from the NCG, four of the Phase 1 reference areas were selected by USEPA for use in Phase 2 sampling to support the BERA and BHHRA (see Figure 2-2 and Figures 2-13 through 2-16). The remaining 10 Phase 1 reference areas were retained to provide data on the bulk surface sediment and surface water characteristics for potential use in understanding regional conditions outside the Study Area.

Reference area selection consisted of a two-step process (see the Phase 2 RI Work Plan Volume 1 [Anchor QEA 2014a] and a presentation to the NCG by USEPA in April 2014 [USEPA 2014b]). The first step involved development of a scoring matrix that documented physical, chemical, and biological conditions in the Phase 1 reference areas and in Newtown Creek. The second step involved evaluation of COPEC concentrations using a variety of screening values. Based on the outcome of the second step, combined with information from the first step, USEPA selected the following reference areas for the Phase 2 program to support the BHHRA, BERA, and other components of the RI Report (e.g., wet and dry weather surface water chemistry) (USEPA 2014b):

- Westchester Creek (Industrial/CSO)
- Head of Bay (Industrial/Non-CSO)
- Spring Creek (Non-Industrial/CSO)
- Gerritsen Creek (Non-Industrial/Non-CSO)

The locations of the four Phase 2 reference areas in relation to Newtown Creek are shown in Figure 2-2. Additional information regarding the sampling conducted at the Phase 1 reference areas and Phase 2 reference areas is provided in Sections 2.1.3 through 2.1.7.

2.1.3 Sediment

The RI and FS Part 1 sediment program included sampling and analysis of surface sediment, subsurface sediment, native material, and sediment traps. In addition, cores were evaluated for the presence of NAPL, and erosion rate (Sedflume) studies were conducted. These studies are briefly described in Sections 2.1.3.1 through 2.1.3.5 and are summarized in Table 2-2b. Study Area sediment sampling stations are presented in Figures 2-17a through 2-17d.

2.1.3.1 Surface Sediment

Surface sediment samples (i.e., samples collected within the top 15 centimeters [cm] [6 inches] of the sediment column) were collected throughout the Study Area, Phase 1 reference areas, and Phase 2 reference areas (see Table 2-2b and Figures 2-3 through 2-17d). These samples were utilized to characterize the chemical and physical nature of surface sediment; provide ecological and human health exposure information; characterize the potential for future natural recovery; provide supporting information for the evaluation of CFT at the groundwater/surface water interface; support gas ebullition evaluations; and support the evaluation of potential remedial alternatives in the FS. Between 2012 and 2017, 453 surface sediment samples were collected from the Study Area, along with 40 samples from the Phase 2 reference areas and 103 samples from the Phase 1 reference areas. Surface water sampling, water quality monitoring, porewater sampling, benthic community surveys, toxicity testing, and bioaccumulation testing were conducted coincidently with select surface sediment samples and are discussed later in this RI Report.

Additionally, 59 shoreline sediment samples (i.e., within the top 60 cm [2 feet] of the shoreline sediment) were collected throughout the Study Area during Part 1 of the FS. These samples were collected to further characterize contaminant distributions in (and near) potentially erodible shorelines to develop and screen remedial alternatives in the FS, as well as to identify potentially significant contaminant sources.

2.1.3.2 Subsurface Sediment and Native Material

Subsurface sediment (i.e., from 15 cm [6 inches] below the sediment surface to the native material interface) and native material sampling was conducted throughout the Study Area to characterize the chemical and physical nature of the sediment; to characterize the vertical distribution and extent of contamination; to inform the groundwater and CFT evaluations; and to support evaluation of potential remedial alternatives in the FS. Subsurface sediment and native material cores and borings were collected from a total of 300 stations in the Study Area between 2012 and 2018, with analysis of 1,224 samples (see Table 2-2b and Figures 2-17a through 2-17d). No subsurface sediment or native material samples were collected from the Phase 1 reference areas or the Phase 2 reference areas. Subsurface sediment temperature was additionally measured over 1 month at two stations in the Study Area to support gas ebullition evaluations.

2.1.3.3 NAPL

The presence or absence of NAPL was evaluated to support the characterization and delineation of NAPL in Study Area sediment and native material. The presence or absence of NAPL was evaluated using a two-step process, where visual observations of potential NAPL in cores were recorded during processing, and the visual observations of potential NAPL were confirmed by shake tests, as described in the Phase 2 RI Work Plan Volume 2 (Anchor QEA 2014b). At least one or more aliquot of sediment from each core was subject to a shake test in the field. Phase 2 sampling was conducted to address data gaps and collect data to support the NAPL evaluation, whereas Part 1 of the FS sampling was conducted to refine the delineation of NAPL. Cores were processed using the two-step process from a total of 169 stations, including all cores and borings collected in Phase 2 and Part 1 of the FS NAPL distribution refinement program; and select archived cores collected in Phase 1 (see Table 2-2b and Figures 2-17a through 2-17d).

NAPL mobility sampling was performed to evaluate NAPL mobility and other physical parameters to develop and screen remedial alternatives in the FS. NAPL mobility evaluation cores collected in Part 1 of the FS were collected from a total of 11 stations located within CM 0 – 2.

2.1.3.4 In-Creek Sediment Traps

The in-creek sediment trap program was conducted to measure gross chemical and solids deposition fluxes and characterize the chemical constituents of newly deposited sediment in the Study Area (e.g., point sources, resuspended solids, East River solids, overland flow). Thirty stations were sampled quarterly during a 9-month period; physical testing was conducted on 134 samples, and chemical testing was conducted on 87 samples (see Table 2-2b and Figures 2-17a through 2-17d).

2.1.3.5 Sedflume

Sedflume sampling was conducted at five Study Area locations in 2012 to evaluate sediment stability and support development of the sediment transport model (see Table 2-2b and Figures 2-17a through 2-17d).

2.1.4 Water

The RI and FS Part 1 programs that evaluated water included surface water sampling, surface water quality profiling, current meters, surface water elevation studies, porewater sampling, and groundwater sampling, as well as point sources, seeps, and overland flow sampling. These programs are briefly described in the following subsections and are summarized in Table 2-2c. Study Area water sampling and measurement stations are presented in Figures 2-18a through 2-18d.

2.1.4.1 Surface Water and Water Quality Profiling

Surface water sampling and water quality profiling were conducted throughout the Study Area, in the Phase 1 and Phase 2 reference areas, and in the East River (see Table 2-2c and Figures 2-18a through 2-18d). The purpose was to characterize surface water conditions, support the BERA and BHHRA, support fate and transport evaluations, and support gas ebullition evaluations. Between 2012 and 2017, 906 surface water samples were collected in the Study Area, 71 in the Phase 1 and Phase 2 reference areas, and 87 in the East River. Sampling was conducted mainly during dry weather periods for Phase 1 reference area and East River programs (though sampling was monthly and dry weather not specifically targeted); Phase 2 included separate wet weather and dry weather sampling programs for Study Area locations.

In conjunction with surface water sampling, vertical water quality profiling was conducted using an in situ meter at each station during the time of sample collection. In addition, water quality profiles were measured during sampling of other media in several other programs, including benthic community surveys, fish and crab surveys, caged bivalve sampling, field gas ebullition surveys (FESs), and at long-term water quality monitoring locations. A standalone tidal survey consisting of collecting water quality profiles during a complete tidal cycle was conducted at three locations in July 2012. In total, approximately 650 vertical water quality profiles were taken within the Study Area.

2.1.4.2 Current Meters

Current meters were deployed in the Study Area to record current velocities at various points within the water column, including devices situated near the sediment bed to support the evaluation of sediment and chemical transport, to provide data related to vessel movement and resuspension of sediment by propwash, and to support the hydrodynamic and sediment transport modeling effort. Bottom-mounted acoustic Doppler current profilers (ADCPs) were deployed in the Study Area in 2012 and 2013 at five stations; and ADCPs and acoustic Doppler velocimeters were deployed in 2015 at six stations each. In addition, boat-based surveys were performed in 2012 and 2013 with ADCPs to build velocity profiles along five cross-creek transects (see Table 2-2c and Figures 2-18a through 2-18d).

2.1.4.3 Surface Water Elevations

The surface water elevation investigation, consisting of the installation of tidal gauges, was completed with the purpose of recording continuous water surface elevations during the RI and Part 1 of the FS to support the hydrodynamic model and better understand water circulation patterns within Newtown Creek. The hydrodynamic model predicts temporal and spatial variation in water depth, current velocity, temperature, and salinity within Newtown Creek. Two tidal gauges were installed in the fall of 2011, and water surface elevations at these locations were recorded during sampling events through May 2018 (see Figures 2-18a through 2-18d).

2.1.4.4 Porewater

Porewater sampling was conducted to provide information to support the BERA, the CFT modeling effort (i.e., to understand sediment-porewater partitioning characteristics and to understand groundwater/surface water interactions), the development of the CSM, and the gas ebullition evaluations. Fifty-nine Study Area and 24 reference area stations were sampled in Phase 2 using passive (solid-phase microextraction [SPME] and peepers) and active (low-flow pumping) methods for porewater analysis either ex situ (in a laboratory coincident with toxicity testing to provide information to support the BERA and gas ebullition investigation) or in situ (at multiple depths within the sediment to support the groundwater investigation) (see Table 2-2c and Figures 2-18a through 2-18d).

2.1.4.5 Groundwater

Groundwater sampling and investigations were conducted as specified in the USEPA *Final Groundwater Investigation Work Plan* (USEPA 2014a) and as a portion of Part 1 of the FS to quantify groundwater discharge to subsurface sediment in the Study Area and identify chemical concentrations associated with groundwater to support the modeling effort. These programs included collection of 65 groundwater samples from the native material underlying the sediment using active (low-flow pumping) methods and hydrogeological data collection, which included use of a hydraulic profiling tool at 24 stations; installation of 14 upland and 13 in-creek long-term monitoring wells for collection of potentiometric data; tidal data and slug testing at 25 stations; and collection of seepage rate measurements at 35 stations during the course of multiple tidal cycles performed by the U.S. Geological Survey (USGS) (see Table 2-2c and Figures 2-18a through 2-18d).

2.1.4.6 Point Sources, Seeps, and Overland Flow

The point sources, seeps, and overland flow programs were conducted to evaluate the composition of potentially significant point and non-point source discharges to the Study Area in support of CSM development, risk assessment, modeling efforts, and eventual evaluation of remedial alternatives in the FS (see Table 2-2c and Figures 2-18a through 2-18d). Point sources, seeps, and overland flow sampling occurred opportunistically and during dry and wet weather events. Opportunistic point source or overland flow samples were collected during the Phase 1 shoreline survey, if point source or overland flow discharges with adequate flow for sampling were observed. Opportunistic sampling during Phase 1 was performed at seven outfalls and pipes observed to be discharging during the Phase 1 shoreline survey. Point sources and overland flow samples were also collected from each type of point source discharge to the Study Area during Phase 2 (i.e., CSOs, stormwater [including overland flow], WWTP treated effluent overflow, and treated groundwater effluent from remediation and dewatering systems). Ninety-six point source discharge samples were collected from the Study Area during Phase 2 using a variety of techniques, including whole-water, bulk-water, WWTP influent, and discrete total suspended solids (TSS) sampling. Seep samples were collected during the shoreline sediment program (Part 1 of the FS) if shoreline seeps were observed on potentially erodible shorelines. Nine opportunistic seep samples were collected from three potentially erodible shorelines (Part 1 of the FS).

2.1.5 Ecological Studies

The ecological studies performed as part of the RI included caged bivalve sampling, benthic toxicity bioassays, benthic community survey and bioaccumulation sampling, fish and crab community survey and sampling, a habitat survey, and a wildlife survey. These studies are briefly described in the following subsections and are summarized in Table 2-2d. Study Area ecological sampling stations are presented in Figures 2-19a through 2-19d.

2.1.5.1 Caged Bivalves

A caged bivalve bioaccumulation study was conducted in 2014 as part of the BERA. The ribbed mussels (*Geukensia demissa*) were deployed at 10 Study Area locations for approximately 60 days, after which chemical concentrations in tissues were measured (see Table 2-2d and Figures 2-19a through 2-19d). Due to the high mortality at one location, the caged bivalves were retrieved 1 month early and analyzed. Water quality profiling was conducted prior to deployment, during monitoring, and during retrieval of the caged bivalves at each sampling station, as described in Section 2.1.4.1.

2.1.5.2 Benthic Toxicity Bioassays

Surface sediment samples were collected for toxicity testing in spring 2014 from 36 Study Area stations and 24 Phase 2 reference area stations (6 stations in each of the four Phase 2 reference areas; see Table 2-2d and Figures 2-13 through 2-16 [reference areas] and Figures 2-17a through 2-17d [Study Area]). Envirosystems, Inc., conducted two tests using the amphipod *Leptocheirus plumulosus*—one to evaluate 10-day acute survival and another to evaluate 28-day chronic survival, growth (as biomass and weight), and reproduction (per surviving amphipod and per surviving female amphipod). To accommodate the large number of samples, each of the two tests were run in two batches; each batch included all 24 Phase 2 reference area samples and 18 of the 36 Study Area samples. Test results for the reference areas were used to develop a reference envelope against which the Study Area results were compared. Porewater chemistry data were collected in sacrificial replicates for the 28-day test using mini-peepers for metals and SPME for polycyclic aromatic hydrocarbons (PAHs), PCB congeners, and pesticides to provide direct measures of exposure to benthic macroinvertebrates and support interpretation of the toxicity test results. Bulk sediment simultaneously extracted metals (SEM) and acid volatile sulfide (AVS) data were also collected at the beginning and the end of the 28-day test to provide additional information regarding metals availability by calculating the sum of SEM minus AVS (Σ SEM – AVS).

2.1.5.3 Benthic Community and Bioaccumulation

The benthic community surveys were performed in 2012 and 2014 to evaluate the Study Area and Phase 2 reference area benthic communities as part of the BERA (see Table 2-2d and Figures 2-13 through 2-16 [reference areas] and Figures 2-19a through 2-19d [Study Area]). Surveys were conducted in the spring (April to May 2012 and May to June 2014), when dissolved oxygen (DO) concentrations are generally near saturation; and in the summer (August 2012 and 2014), when DO concentrations are generally lower in much of the Study Area. In 2012, 34 Study Area stations were surveyed in the spring and summer. In 2014, 56 Study Area stations were surveyed in the spring, and 28 were surveyed in the summer. Additionally, a laboratory-based bioaccumulation study was conducted using sediment collected from 13 of the 2014 Study Area stations, and the polychaete bristle worm (*Nereis virens*). For the four Phase 2 reference areas, 32 stations were sampled in the spring, and 8 were sampled in the summer of

2014. Water column profiling and surface water analytical chemistry sampling was performed at each benthic station, as described in Section 2.1.4.1.

2.1.5.4 Fish and Crab

The fish and crab community surveys and tissue sampling were conducted to support exposure assessments in the BERA and BHHRA and to provide information on community structure for the BERA. In 2012, fish and crab community surveys were conducted in five zones within the Study Area. In 2014, fish and crab community surveys, as well as tissue sampling, were conducted in six zones (fish sampling zones [FSZs]) within the Study Area and in each of the four Phase 2 reference areas (see Table 2-2d and Figures 2-13 through 2-16 [reference areas] and Figures 2-19a through 2-19d [Study Area]).[24] Water quality profiles were measured in each of the fish and crab collection zones prior to sample collection, as described in Section 2.1.4.1.

2.1.5.5 Habitat

Habitat surveys were performed in the Study Area and the Phase 2 reference areas to investigate wildlife habitat conditions for the BERA. Surveys of the Study Area were performed in 2012, and surveys of the four Phase 2 reference areas were performed in 2014, coincident with the wildlife surveys (see Section 2.1.5.6). The land- and boat-based surveys documented plant species; the presence, absence, and condition of shoreline vegetation; canopy type; the presence of upland or wetland communities (if any); the presence and absence of bulkheads; and the presence of any submerged aquatic vegetation.

2.1.5.6 Wildlife

Wildlife surveys were performed in the Study Area and the Phase 2 reference areas to provide information on the presence and activity of wildlife species for the BERA. The wildlife surveys were conducted in 2014 and documented species presence or absence; general frequency of occurrence; site use; foraging activity; and prey type.

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[24] The five 2012 fish and crab community survey zones in the Study Area consisted of two in the main stem of Newtown Creek, one in Dutch Kills, one that included the Turning Basin and Maspeth Creek, and one that included East Branch and English Kills. In 2014, the Turning Basin – Maspeth Creek zone was split into two based on differences in habitat characteristics, for a total of six zones.

2.1.6 Air

Air sampling was conducted along the Study Area and at background locations in the vicinity of the Study Area, to measure the level of breathing zone ambient concentrations and estimate the portion of the measured concentrations that could be potentially attributable to the Study Area. Five background stations were located both upwind and downwind of the Study Area to provide representative background data within the residential and commercial zones surrounding the Study Area. Air sampling of the background stations and 24 Study Area shoreline and in-creek stations was conducted over a 24-hour period in the summer of 2012. The air sampling program is summarized in Table 2-2e, and air sampling stations are presented in Figure 2-20.

2.1.7 Gas Ebullition

Field surveys of gas ebullition were conducted in August 2015 and September 2016 to observe and document the evidence of apparent gas ebullition in surface water, document surface water sheens, and develop an understanding of conditions under which gas ebullition is most likely to occur. Surface water quality profiles were collected from pre- and post-survey baseline stations and from within apparent gas ebullition observation areas, as described in Section 2.1.4.1. Additionally, a pilot study quantitative gas ebullition program was conducted in September 2017 to develop and test methodologies for the FS gas ebullition field program.[25] As part of the pilot study, sediment, surface water, and porewater samples were collected, and sediment temperature and surface water quality profiles were collected from pilot study stations as described in Sections 2.1.3 and 2.1.4. The gas ebullition field survey and pilot study programs are summarized in Table 2-2f and shown in Figures 2-21a through 2-21c.

2.2 Other Studies

Data collected outside of the Newtown Creek RI/FS process were used to supplement the RI depending on the acceptance criteria detailed in the draft DAR (see Appendix J) and approved *Data Management Plan* (Anchor QEA 2011d). Data acceptance criteria include

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[25] Techniques for quantitatively measuring upward fluxes of gas and ebullition-facilitated NAPL/contaminants from the mudline to the overlying surface water are experimental in nature and have not been widely deployed in previous field studies at other CERCLA sites.

minimum data acceptance criteria (MDAC), activity-specific acceptance criteria (ASAC), and related quality assurance codes. MDAC have been developed to determine if historical data are acceptable for use in the RI, without additional review and signoff; this includes data provided from a known source with adequate quality assurance documentation and backup laboratory analytical data. Data not meeting MDAC underwent further review by the Project Manager, RI Manager, or their designees for alternate acceptance for use. Based on their review, the data may be accepted for only limited use, accepted for broader use subject to ASAC, or rejected for all uses. ASAC are need-specific criteria (e.g., age of data, detection limits, and methods use) as appropriate for a specific data use. They are developed for specific technical uses and may include required (must exist for use) and desired (beneficial but not essential for use) criteria. Additional information on data acceptance criteria is provided in the *Data Management Plan* (Anchor QEA 2011d).

One dataset, the 2009 and 2010 National Grid sediment sampling (GEI 2009, 2010), provided a significant number of sediment samples in an area of the Turning Basin and is discussed in some detail in the following subsection. Other non-RI/FS datasets are discussed in the subsequent subsection. Acceptable RI/FS and non-RI/FS data are summarized in Table 2-1. Data completeness for RI and Part 1 FS studies is summarized in Tables 2-3a and 2-3b.

2.2.1 National Grid

GEI Consultants, Inc., on behalf of National Grid, conducted sampling at the Greenpoint Energy Center (DAR No. 32), a former manufactured gas plant (MGP) located adjacent to a portion of the Turning Basin. As part of work related to an uplands investigation, 31 surface and 196 subsurface sediment samples were collected in the Turning Basin of the Study Area in 2010 (GEI 2009, 2010) under a work plan approved by NYSDEC. These data were reviewed for quality and appropriate usage through the DQO process and application of MDAC, ASAC, and related quality assurance codes, as reported in the draft DAR (see Appendix J). The sediment data from the National Grid study were assessed and deemed acceptable by USEPA for inclusion in the RI, as documented in Appendix V to the Phase 2 RI Work Plan Volume 2 (Anchor QEA 2014b).

2.2.2 Additional Data

Information from several other studies is used to support the modeling efforts, groundwater evaluations, source evaluations, BERA, and BHHRA. These data are not in the RI database, as they were either not assessed for MDAC as defined in the draft DAR (see Appendix J) or they only met the criteria for ASAC. This includes regional surface water, sediment, tissue, groundwater, soil, and air data collected by the National Oceanic and Atmospheric Administration, NYCDEP, NYSDEC Contamination Assessment and Reduction Project (CARP), NYSDEC, and various independent parties, as summarized in Table 2-1. When used, in most cases as a supplementary LOE, additional studies are specifically described in evaluations contained in the RI Report appendices.

2.3 Summary of Applicable Site Data and Data Quality Objectives

In total, more than 4,350 samples and more than 1,000,000 individual analytical measurements from more than 900 locations have been collected within the Study Area and the nearby region during the RI and Part 1 of the FS. These data and data from the National Grid investigations were used to support the development of the RI Report. The data used in the RI underwent data quality review and validation and have been determined to be acceptable based on QAPP acceptance criteria. Appendix B provides the data usability assessments, including impacts to DQOs. Rejected data from Phase 1, Phase 2, and Part 1 of the FS are summarized in Appendix Biii Table Biii1-1, Appendix Bi Table Bi2-2, and Appendix Bii Table Bii2-2, respectively.

Additionally, the data were reviewed against the DQOs outlined in the RI/FS Work Plan (AECOM 2011), Phase 2 RI Work Plan Volumes 1 and 2 (Anchor QEA 2014a, 2014b), and FS FP Work Plan (Anchor QEA 2017a) to verify that the completed programs met the goals for the investigations and provide sufficient information to support project decision-making. The DQO process is summarized in the Phase 2 RI Work Plan Volume 1 and includes defining the type, quality, quantity, purpose, and intended uses of data that are being used in the RI (USEPA 2006a). The RI and Part 1 of the FS data meet the DQOs as detailed in Tables 2-2a through 2-2f and are of sufficient quality to update the CSM and complete the RI.

3 ENVIRONMENTAL SETTING

The Study Area and the surrounding watershed have a long history of extensive urban and industrial development dating back to the early 1800s. This has resulted in shoreline and drainage characteristics that are unique compared with other urban waterbodies (Goodwin and Associates 2012). As a result of almost 200 years of industrial, commercial, and residential development, almost all natural stream flow to the Study Area has been eliminated. Instead, watershed drainage is dominated by CSO discharges, more than 300 point source discharges (i.e., WWTP treated effluent overflows, stormwater [including overland flow], treated effluent from groundwater remediation and dewatering systems), and occasional illegal dumping. In addition, groundwater within the recharge area discharges to the Study Area through seeps. The Study Area primarily behaves as a dead-end tidal channel, with flows and currents driven by tides and exchange with the East River. The interplay of these inputs, tidal exchange, the use of this system mainly for marine transport, and the historical discharges from industrial and municipal sources determines the conditions in the creek today.

The Study Area is a highly engineered waterbody that was almost entirely bulkheaded by the early 1900s. Congressionally authorized navigation channels were largely in place by 1910 to support the extension of commercial and industrial traffic. Over time, the types of vessels and navigational uses have changed. The tributaries, except for Whale Creek, are no longer navigable to their authorized depth, due to damaged infrastructure, the presence of subsurface utility corridors at elevations above the authorized depths, shoaling that has resulted from deposition of solids, or construction of low railroad and vehicle bridges that prevent marine vessel access. Nonetheless, there is ship and barge traffic in the main stem and in the lower portion of English Kills, whereas Whale Creek remains an active waterway, used primarily by NYC for operations relating to wastewater treatment.

The urban and industrial nature of the system is a critical consideration of the RI, affecting the assessment of ecological risk and human health risk and the characterization of biological reference conditions. These unique site characteristics and the environmental setting of the Study Area are described in Sections 3.1 and 3.2.

3.1 Physical Environment

3.1.1 Geology and Hydrogeology

3.1.1.1 Geology

The regional geological setting for western Long Island is characterized by Precambrian and Paleozoic crystalline bedrock overlain unconformably by Upper Cretaceous formations (Raritan Clay member, Lloyd sand member, and undifferentiated Matawan Group-Magothy Formation), Pleistocene deposits (Jameco Gravel, Gardiners Clay, and upper Pleistocene glacial deposits), and Holocene post-glacial deposits (salt marsh, stream alluvium, and shoreline deposits). The bedrock crops out in northwestern Queens County and dips toward the southeast. The overlying unconsolidated deposits dip toward the south.

The Study Area is underlain by (from bottom to top) bedrock, Raritan Clay, Gardiners Clay, and relatively permeable native materials, including Upper Glacial Aquifer (UGA) materials and post-glacial deposits (see Figure 3-1). The UGA and bedrock are spatially continuous throughout the Study Area. The Raritan Clay and Gardiners Clay units are mostly present beneath the southern half of the Study Area, and the UGA is underlain by bedrock in the northern half of the Study Area.

The UGA in the vicinity of the Study Area consists of moraine deposits, which are an unsorted and unstratified mixture of clay, sand, gravel, and boulders that were deposited at the front of a continental glacier or at the base of the ice sheet during periods of melting (Buxton and Shernoff 1999).

Upland surface soil overlying the UGA consists of post-glacial deposits and fill. Post-glacial deposits consist of historical marsh deposits, fluvial creek deposits, and lacustrine deposits. The historical marsh deposits correspond to the natural salt marsh deposits that existed along much of the shoreline and the low-lying areas adjacent to the Study Area prior to development and industrialization. The marsh deposits primarily consist of soft to medium soft, gray, black, and brown, organic silty clay, and may act as semi-confining units to the underlying UGA. Fluvial creek deposits mainly consist of compact, gray, black, and brown, organic silty fine to medium sand grading to silty clay. The lacustrine deposits mainly consist of dark gray to brown and olive brown, fine to coarse-grained sand. Fill materials were placed during industrialization of the area, and are highly heterogeneous, including a wide range of materials such as sand, silt, gravel, ash, bricks, wood, and other types of debris, which likely originated both within and outside the Study Area. See Sections 3.2, 3.2.1, 3.2.6, and 3.2.6.21 for more information regarding filling activities within and adjacent to the Study Area.

3.1.1.2 Hydrogeology

Regional hydrogeological units (from bottom to top) include bedrock, Lloyd Aquifer, Raritan Clay, Magothy Aquifer, Jameco Aquifer, Gardiners Clay, and UGA. The Lloyd Aquifer, Magothy Aquifer, Jameco Aquifer, and UGA are regional aquifers. The Raritan Clay and Gardiners Clay units are regional confining units. The bedrock underlying the regional aquifers is composed of crystalline metamorphic and igneous rocks and is considered relatively impermeable (Misut and Monti 1999; Buxton and Shernoff 1999). A modeling study for the WWTP facility (Greeley and Hansen et al. 2000) simulated the horizontal hydraulic conductivity for decomposed bedrock as 0.15 foot per day, which is several orders of magnitude lower than regional aquifers.

In the vicinity of the Study Area, the Magothy Aquifer, Lloyd Aquifer, and Jameco Aquifer are completely absent. Groundwater within the Study Area occurs in three water-bearing units—fill, post-glacial deposits, and the UGA. The UGA is the most transmissive water-bearing unit among the three and is under unconfined or semi-confined conditions. Although the post-glacial deposits and fill units also transmit groundwater under unconfined conditions, the transmissivity of these units is generally considerably lower than the UGA. While the UGA is continuous across the Study Area, the lateral extent of the post-glacial deposits and fill below the water table is limited to the areas that historically were channels or marshes.

Sources of groundwater include recharge from precipitation and artificial returns. Artificial returns are defined as the water that is returned to groundwater as leakage from artificial structures such as water-supply lines and sewer lines, as well as infiltration of irrigation water. Sinks for groundwater include groundwater withdrawals (for remediation, dewatering, or industrial use), losses to sewer lines, leakage to underlying units, discharge to the Study Area, and discharge to the East River. Historically, groundwater in Kings and Queens counties was used as a source of potable water. In the late nineteenth and early twentieth centuries, increasing withdrawals for public supply and industrial uses coincided with the expansion of the municipal sewer system. Instead of recharging the aquifer, groundwater pumped for use in homes and businesses was typically discharged to the sewer system. As recharge decreased and groundwater withdrawals continued, the water table was depressed, and saltwater intruded into the UGA. Withdrawal for public supply and industrial uses declined in the second half of the twentieth century, due to increased salinity in groundwater and the availability of alternative sources of potable water (i.e., water tunnels that conveyed surface water from upstate reservoirs into the city; Cartwright 2002). Currently, there are no permitted potable groundwater sources near the Study Area (NYCDEP 2012a; Plache 2015). Groundwater withdrawals are described in detail in Section 4 of Appendix F.

Groundwater flow in the UGA is generally toward the Study Area and the East River. Within certain portions of the Study Area, groundwater discharges to (i.e., flows into) the Study Area; in other portions, groundwater is recharged by surface water (i.e., flow is from the Study Area into the underlying and adjacent geological formations). Groundwater extraction or dewatering systems near the central portion of Newtown Creek produce localized downward hydraulic gradients, where groundwater flows away from the Study Area. The influence of groundwater extraction and dewatering systems on regional and local groundwater flow is discussed in more detail in Section 4.5.2.1 of Appendix F.

Groundwater within the fill and post-glacial deposits discharges to the Study Area or groundwater extraction systems on the upland, where present. The vertical hydraulic gradient within these units is generally downward, indicating potential for flow to the underlying UGA. Therefore, groundwater that originates in the fill and post-glacial deposits either discharges laterally directly into the Study Area or flows down into the UGA before discharging into the Study Area.

Groundwater flow through Study Area sediment (and potential contaminant load associated with that flow) is a key consideration for the RI; detailed discussion of groundwater flow and chemical loads is provided in Section 5.2 of this RI Report and Sections 4 and 6 of Appendix F.

3.1.2 Sediment Bed Characteristics

The sediment bed throughout Newtown Creek is characterized as a cohesive (muddy) bed, with varying amounts of fine (clay- or silt-size particles) and coarse (sand-size particles) material. The surface sediment is mostly black to brown in color, is generally soft in nature, and exhibits high moisture content and high organic content. Subsurface sediment tends to be black and medium stiff, with less moisture and higher organic content as compared to surface sediment. The sediment bed upstream of approximately CM 2+ (i.e., upstream of the vicinity of the Kosciuszko Bridge) typically contains a mixture of fine and coarse particles with high total organic carbon (TOC) content (often exceeding 10 weight percent [wt%] of the dry sediment mass). The sediment bed downstream of CM 2+ is mainly composed of fine particles with lower TOC content (in the range of 3 wt% to 5 wt% approaching the mouth of the creek). The thickness of the sediment varies throughout the Study Area, due to natural variation in the elevation of the underlying native material (as well as anthropogenic modifications to the creek, including propwash and historical navigational dredging). The sediment is 10 feet thick on average, with a maximum up to 42 feet thick at the confluence of Newtown Creek (mouth) and the East River and up to 33 feet thick in some parts of the tributaries (see Figure 3-2). The sediment is underlain by native materials, which consist of glacial (UGA) and post-glacial (historical marsh, lacustrine, and fluvial creek deposits) deposits (see Sections 4.2.2, 4.3.2, and 4.4.2 for details on the grain size and TOC content of the surface sediment, subsurface sediment, and native material, respectively).

The dynamic nature and history of the Study Area (e.g., filling and bulkheading along the shoreline; changing industrial uses in the watershed; historical and ongoing discharges to the Study Area; navigational dredging; and propwash) have had an influence on the sediment bed characteristics, and this complicates the interpretation of the nature and extent of contamination (discussed further in Section 4).

3.1.3 Bathymetry

Various bathymetric surveys have been conducted in Newtown Creek between 1991 and 2015. Specifically, bathymetric surveys were conducted during 1991, 1999, 2009, 2011 (two surveys), 2012 (two surveys), 2014, and 2015. There were three additional bathymetric surveys completed in the federal navigation channel by the U.S. Army Corps of Engineers

(USACE) in 2016, 2017, and 2019, but these surveys have not been used for purposes of the RI because they were focused on the navigation channel and only covered portions of the Study Area and/or were of lower resolution. The surveys were conducted for different purposes by different entities using various technologies. For example, the 1991 survey was conducted by the USACE using single-beam technology to determine sediment elevations throughout the entire creek and its tributaries. In 1999, the City of New York conducted a single-beam survey to evaluate bathymetric conditions in the tributaries adjacent to their CSO and MS4 discharges. Bathymetric survey information was collected throughout the Study Area in October and November 2011 (single-beam) and again in December 2012 (multi-beam supplemented in some areas by single-beam) to provide a representative dataset for establishing current conditions for the Study Area.[26] This dataset was updated to incorporate the bathymetric survey conducted by NYC in certain portions of the Study Area (i.e., from the mouth of Newtown Creek up to and including portions of Whale Creek) following navigational dredging performed in 2014 (NYCDEP 2014a).

The average bathymetric elevation in the Study Area is -16 feet in North American Vertical Datum of 1988 (NAVD88), with a minimum elevation of approximately -31 feet in NAVD88. Water depths in the Newtown Creek main channel extend up to approximately 28 feet below mean lower low water (MLLW), but generally are in the range of 18 to 20 feet within the center of the channel (MLLW is +0.261 foot above NAVD88). Within the Turning Basin, water depths range between approximately 12 and 24 feet below MLLW. The tributaries exhibit a great deal of variability in water depths and are generally shallower than the main stem; these shallower depths are a result of alterations over the last century associated with reduced (or no further) navigational use and ongoing deposition of solids. Figure 3-3 depicts the bathymetry elevations of the site.

3.1.3.1 Debris

In 2011, a search for submerged utilities and infrastructure was conducted to refine sediment sampling locations and to inform future investigation and potential remedy evaluation. An

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[26] The 2012 survey was conducted following Hurricane Sandy, which caused extensive flooding throughout the region. Comparison of pre- and post-storm bathymetry showed no significant elevation change that could be detected, beyond the accuracy of the data collected.

extensive hydrographic survey program, including a side-scan sonar and magnetometer survey, was conducted to evaluate the location and nature of obstructions present (Anchor QEA 2012a). The side-scan sonar survey identified anthropogenic debris scattered throughout the survey area, consisting of automobiles, linear objects resembling structural beams, derelict piles, tires, ladders, concrete construction blocks, slumped riprap, and numerous other anomalies. Details of the side-scan surveys can be found in Section 2.1.3 of the Phase 1 IDR (see Attachment Bi-A of the Phase 2 DSR, which is included in Appendix Bi to this RI Report), with results of the surveys presented in the Phase 1 IDR Figures 2-2a through 2-2j.

Magnetometer survey data were collected to differentiate metallic debris from other large surface debris identified during the side-scan sonar survey, to identify large buried metallic debris not visible from the side-scan sonar survey, and potentially to locate buried utility crossings within the survey area. The data suggested widespread distribution of anomalies throughout the survey area, but interpretation is challenging due to the urban nature of the Study Area shoreline. Due to the magnetic interference from the surrounding urban environment, there is low confidence that the digitized magnetic anomalies represent only surficial or buried debris within the Study Area.

3.1.4 Hydrodynamics

The current understanding of hydrodynamics within the Study Area is well informed by both extensive Phase 1 and Phase 2 data collection and reliable predictions of the point source and hydrodynamic models (additional discussion of hydrodynamics is provided in Section 6.2, and detailed documentation of the hydrodynamic modeling is provided in Section 4 of Appendix G). Newtown Creek receives water from the East River as part of the daily tidal exchange, with the East River tidal volumetric flow into Newtown Creek being approximately 70 times greater than the total point source volumetric discharges on an annual basis. Flows enter Newtown Creek from point source discharges (i.e., CSOs, WWTP treated effluent overflow, stormwater [including overland flow], and treated effluent from groundwater remediation and dewatering systems), and groundwater flow. Figure 3-4 shows the relative percentages of these components in total annual freshwater discharge to the Study Area. Approximately 92% of the total CSO discharge to the Study Area occurs at the four largest outfalls located at the heads of tributaries. CSO and stormwater discharges including overland flow account for approximately 62% of the total freshwater flow to the system, which affects the hydrodynamics of the site during discharge events.[27]

Because Newtown Creek is a dead-end tidal channel with point sources and overland flow as the predominant freshwater inflows, the creek is heavily influenced by the tidal exchange with the East River during non-storm conditions and by the discharges from larger point sources (CSOs and stormwater) under storm conditions. Typical of a dead-end tidal channel, current velocities have a maximum value near the confluence of Newtown Creek (mouth) and East River that decrease with increasing distance from the East River, with relatively stagnant conditions in the upper portions of the Study Area during dry weather (e.g., English Kills). During dry weather conditions (i.e., minimal inflow from point source discharges and overland flow), the hydrodynamics in the creek are primarily affected by tidal circulation, with minimal vertical stratification. Groundwater inflow, where present, does not significantly affect hydrodynamic processes (i.e., circulation, stratification; see Section 4 of Appendix G) based on diagnostic testing with the hydrodynamic model. In contrast, during wet weather conditions, when freshwater is discharged into the Study Area from point sources and overland flow (CSOs being the most significant on a volumetric basis), stratified flow conditions oftentimes develop. Less dense, fresher water flows toward the East River in a surface layer, and denser, more saline water flows inland in a bottom layer. The strongest salinity stratification during a point source discharge event occurs in the tributaries, with salinity stratification occurring to a lesser degree in the Turning Basin and occurring the least between CM 0 and 2 (due to mixing during tidal exchange). Although the influence of point source discharges on stratification dissipates relatively quickly following a storm event, the actual residence time of the water discharged within the Study Area from point sources is longer than stratification dissipation, due to mixing and exchange processes in the dead-end tidal channel (additional discussion of mixing and point source discharge effects based on a 1.5-day tracer simulation is provided in Section 6.2.2).

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[27] Flow estimates are discussed in detail in Section 5.1.2. Annual CSO and stormwater discharge volumes were estimated using the 2015 geo-neutral point source model. Annual discharge volumes for WWTP treated effluent and treated effluent from groundwater remediation and dewatering systems were estimated based on information provided by the owners of the discharges.

3.1.5 Water Quality

Water quality within the Study Area was investigated through the extensive sampling program conducted during Phase 1 and Phase 2 (see Section 2.1.4). That sampling included in situ profiling to measure standard water quality parameters such as salinity and DO, as well as collection of samples for laboratory analysis of TSS and chemical constituents. For the chemical constituents being evaluated in the RI, concentrations in surface water are described in detail in Section 4 (nature and extent), with additional discussion in Section 6 (fate and transport). TSS is included in these discussions in Sections 4 and 6 as well, and it was evaluated extensively as part of the sediment transport modeling effort described in Section 3.1.6 (with detailed evaluations in Section 5 of Appendix G).

During dry weather conditions, salinity in Study Area surface water, which provides a measure of the relative amounts of saline/brackish tidal water from the East River and freshwater from point sources and overland flow and groundwater, generally ranges between 20 and 25 practical salinity units (psu) (which is characterized as polyhaline). Salinity varies based on tidal influence, wet weather flow, and distance from the mouth of Newtown Creek. Data collected as part of the RI surface water program (see Section 4.7) showed that during dry weather conditions, salinity in the Study Area and the East River are similar. During wet weather discharge events, the surface water data showed that salinity stratification occurs in the water column, with a less saline layer (often less than 10 psu) of varying thickness occurring at the surface; the extent and variability of such stratification is greater in the tributaries (Dutch Kills, Maspeth Creek, East Branch, and English Kills) than in the main stem (CM 0–1 and CM 1–2). Density-driven circulation (of varying intensity, depending on tidal conditions and freshwater volume for a specific point source and overland flow discharge event) occurs within the system during the episodic freshwater inflow events, as a result of less-dense, fresher water flowing toward the East River in a surface layer and denser, more saline water flowing inland in a bottom layer.

Surface water quality within the Study Area is influenced by the hydrodynamic characteristics (see Section 3.1.4) and sources of inflow into the creek. As discussed previously, while freshwater flow to Newtown Creek is dominated by point source discharges (NYCDEP 2011a), the effects of freshwater flows from point source discharges and groundwater will be incorporated into the hydrodynamic, sediment transport, and chemical fate models.

Therefore, water quality in Newtown Creek is directly impacted by CSOs, WWTP treated effluent overflow to Whale Creek, stormwater (including overland flow), industrial discharges, inflow from the East River, and sediment (e.g., oxygen demand), and groundwater. Water quality monitoring, data analyses, and mathematical modeling performed by NYCDEP since 1980 have consistently documented anoxic conditions and other surface water and sediment quality impairments throughout the system (LMS 1992; URS 2011; NYCDEP 2011a). Surface water sampling and water quality profiling conducted as part of the RI during wet and dry weather conditions confirmed the influence of point source discharges on water quality in the creek (described in Section 4.7). Impacts from CSOs and the installation of the aeration system in English Kills are discussed in Section 3.2.8.3.1. Surface water classifications and criterion applicable to Newtown Creek are discussed in Section 3.2.9.

3.1.6 Sediment Transport

Current understanding of sediment transport within the Study Area is informed by both Phase 1 and Phase 2 data collection, as well as predictions of the sediment transport model (additional discussion of sediment transport is provided in Section 6.3, and detailed documentation of the sediment transport modeling is provided in Section 5 of Appendix G). The creek is primarily a depositional system, due to the relatively low tidal current velocities. Erosion due to currents is limited and localized (e.g., sediment mounds in the vicinity of CSO discharges in English Kills, East Branch, and Maspeth Creek) based on evaluation of data and modeling. Propwash from ship movement can cause episodic bed scour, mixing, resuspension, and redeposition within the main channel and Turning Basin (see Section 7.3.2 of Appendix G). The overall rates of deposition and erosion are quantified by the net sedimentation rate (NSR), which has been characterized based on extensive data-based and modeling evaluations (see Section 6.3.4 of this RI Report and Section 5.2.1 of Appendix G). The NSR varies spatially within the Study Area and is relatively higher at the mouth near the East River and in the tributaries, with relatively lower NSRs in the Turning Basin. Multiple LOEs from the data evaluations (i.e., geochronology cores, differential bathymetry analysis, historical dredging information, sediment trap data, and vertical profiles of chemical concentrations in the sediment bed) indicate that historical NSRs were much greater (e.g., dating back 20 to 50 years or more) than more recent rates. This temporal decrease in NSRs is primarily due to decreases in CSO and stormwater sediment loads during the last 50 to 75 years. Although limited empirical evidence is available to directly quantify these changes in CSO- and stormwater-derived sediment loads, data from historical CSO sampling (e.g., see Section 4.2.4 of Appendix E), along with operational changes driven by Clean Water Act (CWA) regulation, such as increases in treatment plant capacity and implementation of green infrastructure projects, support this conclusion. Localized decreases in sediment trapping efficiency have also contributed to these observed temporal decreases in NSRs, for example, in the sediment mounds near large CSO outfalls (e.g., see Attachment G-H of Appendix G). Sediment depositing in the system and contributing to observed NSRs originates from two primary sources: point source discharges (which include CSO, stormwater, and permitted discharges—discussed in more detail in Section 5) and the East River. Point source sediment loads occur during episodic discharge events that typically last 2 to 5 hours (see Section 3.9 of Appendix G). Sediment loads from point source discharges tend to have relatively high TOC content and are composed of an approximately even mix of fine and coarse particles (see Table G5-9 of Appendix G). The East River sediment load has a lower TOC content (see Section 4.2.2.3 and Figure 4-8), and that load is primarily composed of fine sediment particles. East River sediment transported into Newtown Creek mixes with solids from point source discharges in the water column, with a portion of the East River sediment load (on an annual basis) being deposited within the Study Area (see Section 5.6.1.2 of Appendix G).

In summary, Newtown Creek is a complex, dynamic aquatic system. Twice-daily tidal flows from the East River interact with storm-driven upstream freshwater inputs in a complex geometry to create dynamic local environments, each of which exhibits a unique combination of solids loads and depositional characteristics. Ship and barge traffic lead to localized sediment resuspension, further impacting the dispersal and deposition of sediment and the chemicals sorbed to sediment (see Section 5.5.3 of Appendix G). In the historical context, sediment and chemical loads to this complex waterbody have changed dramatically over many decades, both in magnitude and in location along the creek (see Section 3.2). Furthermore, the complex surface water mixing, depositional and erosional processes, and groundwater flows have changed over time, due to changes in creek geometry, creek bathymetry, groundwater and storm-related flows, and human activity. Thus, the current distribution of chemicals in the sediment and surface water of Newtown Creek that are observed today, and that are the subject of this RI, reflect nearly two centuries of complex, dynamic loads and fate and transport processes.

3.1.7 Habitat

Situated between Brooklyn and Queens, Newtown Creek and its tributaries provide a unique and locally important shallow water habitat for a variety of species from crabs and fish to birds and mammals (see Section 3.1.8). The aquatic habitat is mostly subtidal, supporting benthic macroinvertebrates (dominated by oligochaetes [segmented worms]), polychaetes (bristle worms; *Nereis virens*), amphipods (small crustaceans), epibenthic invertebrates such as crabs and bivalves, and resident fish such as mummichog (*Fundulus heteroclitus*), as well as limited use by non-resident migratory fish such as striped bass (*Morone saxatilis*). Intertidal areas that could provide additional foraging habitat for fish and crabs, as well as potential forage habitat for birds and mammals, are mostly confined to the headwaters of Maspeth Creek. At low tide, approximately 5% of the Study Area could potentially provide forage habitat for sediment-probing birds or birds that forage by wading in shallow water (see Section 2.1.3 of Appendix I). Of that, 68% is within sediment mounds or "mud flats" located in the vicinity of some active outfalls and bulkheaded areas; the remaining 32% is made up of riprap or narrow strips of shoreline. Due to the semidiurnal tidal patterns in the Study Area, forage habitat for these sediment-probing and wading birds decreases rapidly as the tide rises and is close to 0% at high tide. In contrast, for fish, crabs, and birds such as the double-crested cormorant (*Phalacrocorax auritus*) that forage by swimming and diving, available forage habitat increases during high tide. For mammals such as the raccoon (*Procyon lotor*), access to the intertidal areas is limited in some areas due to the vertical bulkheads and other anthropogenic features along the shoreline, although raccoons may have some access behind and under bulkheads.

Submerged macrophytes were occasionally observed floating in the Study Area near the East River (e.g., sea lettuce [*Ulva lactuca*]), or attached to pilings and riprap (e.g., green algae [*Enteromorphia*] and brown algae [*Desmarestia*]), but none were observed rooted in the Study Area sediment. The only rooted emergent aquatic macrophytes observed during the Phase 2 sampling were patches of non-native common reed (*Phragmites australis*) toward the head of Maspeth Creek. The lack of an aquatic macrophyte community in the Study Area could be due to a number of physical attributes such as limited sloped areas due to the vertical nature of the shoreline, wave action from daily boat and barge traffic, and high turbidity leading to sunlight limitations and reduced photosynthesis, as well as contaminants. These attributes, in combination, likely limit the establishment and growth of macrophytes (see Section 2.1.3 of Appendix I).

Historically, Newtown Creek and its tributaries have been impacted by industrial discharges, municipal discharges, and spills. As described in Section 3.2.6.15, by the late 1880s, numerous oil refineries were operating along the creek, and additional manufacturing operations followed into the beginning of the twentieth century. These included petrochemical plants; animal rendering operations, such as fertilizer and glue factories; copper smelting; sugar refining; pencil manufacturing; manufactured gas production; sawmills; and lumber and coal yards.

Direct discharge to Newtown Creek was the primary method for disposal of stormwater, sewage, and industrial wastewater prior to the early twentieth century. Historical industrial discharges included waste liquor from grease vats and digesters at fertilizer and fat rendering facilities; oily wash water generated during the kerosene treating process at refineries; condensate from exhaust hoods at varnish and printing works; effluent from air-scrubbing systems; and cooling water, plating wastes, and plant clean-up water from the manufacture of electrical wiring devices (BPL 1890; Hazen and Sawyer 1960; Hurley 1994; NYSL 1900; Baker and Kent 1887).

Early municipal sewer systems also discharged directly to the creek. In the late 1800s and the first half of the 1900s, direct discharges from sewers operated by NYC regularly added raw sewage and other pollutants into the creek. By 1910, intercepting sewers (i.e., interceptors) had been constructed in some areas near the creek to convey flows to the East River where they were discharged without treatment. However, stormwater, sewage, and industrial wastewater continued to discharge to the creek via relief outfalls in areas without interceptors and when the capacity of the interceptor pipes was exceeded (Metropolitan Sewerage Commission of New York 1910; Board of Water Commissioners 1867; War Department 1891, 1915, 1936a; USEPA 2011). Since 1967, when the Newtown Creek WWTP began operating, stormwater (from areas served by combined sewer systems), sewage, and industrial wastewater flows have been conveyed to either the Bowery Bay WWTP or Newtown Creek WWTP for treatment prior to discharge outside the Study Area. However, in some portions of the Study Area, direct discharges of stormwater from private sites and MS4s, as well as CSOs, continued and are ongoing today (see Figure 3-5 and Section 3.2.8 for more details on discharges to Newtown Creek).

As discussed in Section 5 of this RI Report and Section 2.1.3.4 of Appendix E, annual CSO discharge to the creek is approximately 1,600 million gallons (MG).[28] Bypasses may also occur during dry weather when the combined sewer infrastructure malfunctions (NYCDEP 2011c, 2012b, 2013a, 2014b, 2015, 2016a). These dry weather bypasses are infrequent, short-term discharges and result in discharge of a small volume compared to that of the wet weather CSO discharges. In areas not served by combined sewers, direct discharges of stormwater from MS4s (330 MG annually) to Newtown Creek are ongoing.

As documented by NYC, CSO discharges have likely contributed to subtidal surface sediment with an organic carbon (OC) content greater than 3 wt%, which may contribute to impairment of the benthic macroinvertebrate community in Newtown Creek (Hyland et al. 2000 as cited in NYCDEP 2011a). The bacterial decomposition of organic matter results in a decrease in DO and an increase of toxic byproducts such as ammonia and sulfide (Diaz and Rosenberg 1995; Hyland et al. 2005; Norton et al. 2002; Pelletier et al. 2011). DO below 2 milligrams per liter (mg/L) results in hypoxic conditions that adversely affect the respiration of benthic macroinvertebrates (CENR 2010; Gray et al. 2002; Brown et al. 2000) and can result in local extinction, except within the microbial community (CENR 2010; Llanso 1992). This is made worse during the summer months when water temperatures are elevated and the bacterial degradation of organic matter is accelerated (Gray et al. 2002). During the summer Phase 1 surveys in 2012, surface water DO at depth fell below the New York State Class SD threshold of 3 mg/L, particularly in the tributaries; the benthic macroinvertebrate community was impaired even further, with no macroinvertebrates found in surface sediment at tributary stations (see Appendix I). A series of infrastructure projects are being completed by NYC in the Newtown Creek sewershed to address CWA

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[28] Annual CSO and stormwater flows for 2008 to 2012 were estimated from historical precipitation using the 2015 geo-neutral point source model. The arithmetic average annual discharge for 2008 to 2012 is reported.

requirements such as surface water DO. These projects include the installation of an aeration system in certain areas of Newtown Creek and a CSO long-term control plan (LTCP; see Section 3.2.8.3.1).

3.1.8 Ecological Community

The benthic macroinvertebrate community (benthic community) of the Study Area has been described as pollution tolerant, exhibiting low species diversity and abundance (NYCDEP 2011a). Dominant taxa from the Phase 1 and Phase 2 benthic community surveys are oligochaetes, polychaetes, and amphipods (see Section 2.1.4 of Appendix I; Attachment Bi-A of Appendix Bi). Overall, the benthic community is considered stressed, based on Phase 1 and Phase 2 benthic community data. This is demonstrated by low scores for the Weisberg Biotic Index (WBI), as a measure of benthic invertebrate health (Adams et al. 1998).

The fish community of the Study Area has been reported to be sparse, especially during the summer months, when DO concentrations can drop below 1 mg/L (NYCDEP 2011a). Based on the Phase 1 and Phase 2 fish and crab community surveys, the dominant fish species of those collected are mummichog, Atlantic menhaden (*Brevoortia tyrannus*), and striped bass (see Section 2.1.4 of Appendix I; Attachment Bi-A of Appendix Bi). The most common species of crab collected in the Study Area is the blue crab (*Callinectes sapidus*), followed by the green crab (*Carcinus maenas*) and rock crab (*Cancer irroratus*). The horseshoe crab (*Limulus polyphemus*) from the subphylum Chelicerata (not a true crab) was also a common species of those collected.

Observations of birds and mammals were made during Phase 1 and Phase 2 to provide a qualitative evaluation of wildlife present in the Study Area. The species observed during the time frames over which the field surveys were conducted (Phase 1, November 2011 and June 2012; Phase 2, May through June 2014 and August through September 2014) are species commonly found in urban environments, such as gulls (*Larus* species), crows (*Corvus brachyrhynchos*)*,* European starlings (*Sturnus vulgaris*)*,* Norway rats (*Rattus norvigicus*), and raccoon (see Section 2.1.4 of Appendix I; Attachment Bi-A of Appendix Bi). Of the semiaquatic birds, the spotted sandpiper (*Actitis macularius*) and double-crested cormorant were frequently observed. Other semiaquatic birds observed included the black-crowned night heron (*Nycticorax nycticorax*), green heron (*Butorides virescens*), great egret (*Ardea alba*), and infrequently, belted kingfisher (*Ceryle alcyon*). No amphibians or reptiles were observed in the Study Area during the Phase 1 or Phase 2 surveys.

3.2 Human Use

Historical records indicate that the Newtown Creek area was used by the Mispat tribe up to the early 1600s. In 1613, the Dutch explored and surveyed Newtown Creek, subsequently acquiring the area from the local Mispat tribe. By the mid-1600s, the area was primarily occupied by farmsteads and small agrarian villages. Through the 1700s, the marshland around Newtown Creek and its tributaries made it a difficult and expensive challenge to maintain serviceable roadways. Because waterway transportation was economic and efficient, passengers and commodities moved primarily by water (Goodwin and Associates 2012). By the mid-1800s, farms and plantations lined the shores of the creek. In 1850, an upland area at the mouth of Newtown Creek known as Hunter's Point was subdivided. Adjacent marshes were filled,[29] and bulkheads were constructed to support commercial and industrial development along the creek (NYSDOT and FHWA 2005).

The Newtown Creek area has a history of extensive urban and industrial development. Modifications to the physical layout of the creek shoreline and configuration of freshwater discharges have resulted in a system that is largely engineered for industrial, municipal, and navigational purposes. Historically, freshwater flow to the creek included two components, tributary flow and groundwater flow. Many decades of urban development have led to the elimination of tributary flows and to the creation of freshwater point source discharges. Dating back to the 1800s and the early 1900s, untreated stormwater, industrial wastewater, and domestic sewage were generally discharged directly to Newtown Creek (City of

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[29] Historical documentation indicates that low-lying areas near the Study Area were filled throughout the nineteenth and early twentieth centuries by numerous private owners and public entities. In the nineteenth century, trash and ashes were used to fill many low-lying areas along the creek, as discussed in Section 3.2.6.21.2. In the early twentieth century, 4 feet of earthen fill was placed in swampy areas north of Dutch Kills during the construction of the Sunnyside Yard (Field Engineer 1918). In the early 1940s, the federal government constructed a wood bulkhead and filled the ground behind it with 290,000 cubic yards of earth fill taken from upland areas of the property (Defense Project Plancor 1942). Although fill materials often originated outside the Study Area, USACE specifications allowed for depositing dredged materials behind bulkheads in shoal waters or at the water's edge (Rostock 1922).

Brooklyn 1897; Metropolitan Sewerage Commission 1910; War Department 1939; USACE and USN 1939; Hazen and Sawyer 1960). The present-day system of intercepting sewers (i.e., interceptors) and the Bowery Bay WWTP and Newtown Creek WWTP were operating by 1967; however, direct discharges of stormwater and CSOs to the Study Area are ongoing today. Types of current point source discharges to the Study Area are CSOs, the Newtown Creek WWTP treated effluent overflow, stormwater (including overland flow), and treated effluent from groundwater remediation and dewatering systems (see Section 3.2.8). Today, Newtown Creek is fully urbanized, with no remaining natural marshlands or freshwater streams. CSOs discharge approximately 1,600 MG[30] of untreated combined flows (i.e., sewage and stormwater), and the Whale Creek WWTP treated effluent overflow contributes an additional 730 MG into the creek every year (Anchor QEA 2014a, 2014b). In areas not served by combined sewers, direct discharges of stormwater from MS4s (330 MG annually) to Newtown Creek are ongoing.

Significant changes have occurred in the use of Newtown Creek and the surrounding uplands since the early 1800s. Industrial activities in the surrounding uplands and use of the creek for shipping and navigational purposes increased steadily after the Civil War. In 1912, the *New York Times* reported that Newtown Creek "has commerce greater than that of the Mississippi River or any of its tributaries" (NYT 1912). Historical industrial operations located around Newtown Creek included adhesives factories; animal rendering, glue factories, and fertilizer plants; asphalt production, mixing, and storage operations; automobile manufacture, repair, and service; canneries; coal processing, handling, and storage; copper wiring plants; creosote production and treatment; distilleries; electronics and electroplating industries; hide-tanning plants; incinerators; MGPs; metal production, smelting, metal works, and fabricating; metal scrap and storage; municipal wastewater treatment; paints and pigments industry; paper products industry; pencil manufacturing; petroleum refining and bulk storage; plastics industry; printing; railyards; sawmills and

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[30] The 2015 version of the geo-neutral model provided by NYCDEP in 2016 was used to estimate flows for the RI load estimates. As discussed in Appendix E, prior to sampling, the 2013 geo-neutral model estimates were used in the evaluation of point source and overland flow discharges and the selection of sampling locations. However, the 2015 version of the geo-neutral point source model provided by NYCDEP was used to estimate flows described in this report. The geo-neutral point source model may be revised or augmented in the future to improve predictions or estimates of freshwater inflow to the Study Area from various discrete subbasins (e.g., direct drainage areas).

lumberyards; shipbuilding; solid waste disposal/landfilling; sugar refining; utilities; and waste oil refining operations (NYSDOT and FHWA 2005; see Appendix J). Following World War II, marine cargo on Newtown Creek decreased significantly and there was a shift away from manufacturing facilities to materials handling facilities (NYSDOT and FHWA 2005). Today, the predominant land use around Newtown Creek and the tributaries remains industrial, with pockets of mixed use, commercial, and residential developments (NYCDEP 2011a). Industrial activities near the creek currently include the following: warehouse and distribution facilities; vehicle storage and maintenance; electrical distribution; plastics and foil manufacturing; waste transfer yards and recycling facilities; road service support facilities; construction materials storage; facilities that store electrical equipment; scrap metal processing facilities; lumberyards; ready-mix concrete plants; bulk fuel distribution terminals; railroads (e.g., tracks, yards); utilities; and municipal wastewater treatment (see Appendix J; Anchor QEA 2014a, 2014b). A summary timeline of varied human use and activities on and near the creek over time is shown in Figure 3-6.

Public health, environmental, and land use regulations were influential in the historical development of the creek and the surrounding uplands and continue to play an important role in the future of Newtown Creek. Documentation of low DO concentrations, elevated bacteria counts, and other water and sediment quality impairments in Newtown Creek resulting from CSOs can be found in public health reports beginning in the early 1900s and continuing through the present day (Metropolitan Sewerage Commission 1910; Hazen and Sawyer 1960; NYSDOH 2014). Today, NYCDEP is responsible for managing CSO discharges to Newtown Creek in accordance with federal CSO policy, the water quality goals of the CWA, and analogous New York State requirements. An overview of current and historical CSO discharges and regulation of CSO discharges to Newtown Creek is presented in the timeline in Figure 3-7.

3.2.1 Creek Configuration

Dredging and filling events and bulkhead construction have occurred in the Study Area, causing changes to the shoreline configuration, from the mid-1800s to the present day. Changes in the historical configuration of Newtown Creek largely coincided with the industrialization of the area during the mid-1800s and early 1900s. The creek once flowed through wetlands and marshes and had a natural depth ranging from 4 to 12 feet (War Department 1916). Historical maps depict Newtown Creek with natural side channels and islands including Mussel Island, which was a small mud flat approximately 3.7 acres in size and located opposite the mouth of Maspeth Creek (MANY 1921). Over time, Newtown Creek and its tributaries were channelized and dredged, and wetlands and marshes were filled, reworking the natural banks and creek into a controlled channel (AECOM 2011). Figures 3-8 through 3-12 depict changes in Newtown Creek's configuration between the mid-1800s and today. The figures indicate that the creek configuration has remained essentially unchanged since the late 1900s.

Early landowners (1850s) contributed to the marsh filling and bulkhead construction. The Long Island Rail Road (LIRR) filled and bulkheaded approximately 2,000 feet along the creek west of Dutch Kills in the 1870s, which contributed directly to shoaling and siltation by reducing channel widths and tidal scouring. In the 1880s, USACE became responsible for channel improvements and maintenance dredging (NYSDOT and FHWA 2005).

Between the late 1880s and the 1920s, the shoreline of the Turning Basin underwent a change. Newtown Creek to the east had previously consisted of two channels that wrapped around Mussel Island. By 1925, this feature was eliminated (i.e., Mussel Island was removed, and the channel between Mussel Island and the mainland was filled in). Additional filling activities are discussed in Section 3.2.6.21.

By 1921, the entire lower 2.5 miles and approximately one-half of the upper 1.5 miles of Newtown Creek waterfront were bulkheaded. Dutch Kills and Whale Creek were bulkheaded throughout approximately one-half of the course, and English Kills was bulkheaded throughout almost the entire length (MANY 1921). Undeveloped sections without bulkheads included Maspeth Creek and parts of Dutch Kills; 3,000 feet of shoreline along the Queens side of Newtown Creek; and 3,500 feet of shoreline along the Brooklyn side of Newtown Creek downstream of Maspeth Avenue. By the 1930s, 96 bulkhead terminals were documented along Newtown Creek, with water depths ranging from 3 to 11 feet MLLW (War Department 1936b).

During the 1990s and 2000s, many property owners replaced deteriorating timber crib bulkheads. The replacements typically included the installation of steel sheetpiles, with or without anchors, seaward of the existing timber cribs, or the addition of riprap to stabilize the shoreline (NYSDOT and FHWA 2005; AECOM 2011).

Today, the Newtown Creek shorelines and associated tributaries are almost entirely bulkheaded. A delineation of bulkhead materials was first documented during a 2011 RI shoreline assessment survey (Anchor QEA 2012a) and updated in 2013 during the wet weather survey conducted as part of the evaluation for other non-point sources (draft *Sources Sampling Approach Memorandum* [SSAM]; Anchor QEA 2014m). Ninety-nine percent of the shoreline contains bulkheads, with approximately 63% that are in good to excellent condition and approximately 37% that are in poor condition. Bulkheads are generally constructed of timber (21% of total bulkheaded length), steel sheetpile (17%), concrete (30%), or riprap (32%). Less than 1% of the shoreline is bare ground. The shoreline survey results are shown in Figure 3-13.

3.2.2 Creek Crossings

The first bridge crossing Newtown Creek was likely a wooden bridge near the present location of Meeker Avenue, which was in place by 1670 (NYSDOT and FHWA 2005). The wooden bridge was replaced between 1812 and 1814 by the Penny Bridge. The first crossing of English Kills was constructed circa 1814 to 1816, along the present location of Metropolitan Avenue as part of the Williamsburg and Jamaica Turnpike. Other nineteenth century-constructed bridges included the Maspeth Avenue Plank Road to Newtown Bridge in 1846 and the Greenpoint and Flushing Plank Road Bridge between 1853 and 1854. Two bridges were constructed as part of local road development in the late 1860s, over Newtown Creek at Vernon Avenue and Dutch Kills on the present location of Borden Avenue. The first railroad crossing over Newtown Creek, a swing bridge, was constructed in 1861. Nine additional bridges were constructed in the twentieth century, including swing, fixed, bascule, and retractile styles (NYSDOT and FHWA 2005). Today, Newtown Creek and its tributaries are crossed by 11 bridges, 9 movable and 2 fixed, including the recently replaced Kosciuszko Bridge.

Numerous structures exist under Newtown Creek, including tunnels, pipelines, and marine cable crossings (see Figure 3-14). The Standard Oil pipeline, associated with the petroleum industry expansion in the area, was constructed under Newtown Creek in 1879. A subway tunnel was constructed in the early twentieth century, under the lower portion of the waterway. A 54-inch-diameter sewer pipe (inverted siphon), crossing the creek at Maspeth Avenue, was constructed in the mid-1930s (War Department 1934, 1936a). Buckeye Pipeline, an underground petroleum pipeline that transmits petroleum products from Linden, New Jersey, across Staten Island, under the New York Harbor and into Brooklyn and Queens, began operating on December 9, 1966 (City of New York Board of Estimate 1967). A 1974 USACE navigability determination found that approximately 15 pipelines, 15 submarine cables, and one tunnel crossed under Newtown Creek at various locations (USACE 1974; NYSDOT and FHWA 2005).

The Con Edison – 11th Street Conduit (DAR No. 110) was constructed in the 1960s and provides an artificial closed conduit under Newtown Creek, which houses the Consolidated Edison Company of New York (Con Edison) electrical distribution cable between Brooklyn and Queens. The tunnel extends from the site at Ash Street and McGuinness Boulevard in Brooklyn to 11th Street and 47th Road in Queens (104[e] Superseding Response Appendix A, provided on December 12, 2012).

In 2011, a search for submerged utilities and infrastructure was conducted to refine sediment sampling locations and to inform future investigation and potential remedy evaluation. Utility crossings included cross-creek features, such as pipes, cables, lines, or wires that appeared to transmit or transport electrical, gas, communication, or other utilities. The search for submerged utility crossings included a Freedom of Information Act request to USACE; surveys of the Study Area; and a utility locate. The search concluded that there are approximately 22 utility crossings in the Study Area, 15 in Newtown Creek, and 7 in the tributaries (electrical cables and pipelines in Dutch Kills, electrical cables in East Branch, and aeration piping and a gas pipeline in English Kills) (Anchor QEA 2012a). Additionally, an aeration piping system was installed in the Turning Basin and East Branch in 2016 (Mechanical and Marine Construction Corp. 2017). Many of these crossings are at shallower depths that do not accommodate dredging to the current federally authorized navigation channel depths.

3.2.3 Historical and Current Shipping Activity

Due to limited road development and bridge crossings on Newtown Creek, water transport was viewed as the best way to transport goods and people among local villages and nearby urban areas through the 1860s (NYSDOT and FHWA 2005).

In the 1850s and 1860s, petroleum refining industries began locating along the creek, further transforming Newtown Creek into an industrial waterway. Shipping vessels brought in crude petroleum and shipped out refined petroleum products such as kerosene (Goodwin and Associates 2012; NYSDOT and FHWA 2005).

Between 1880 and 1900, channel improvements, improved adjacent street networks, and the availability of rail and lighterage service supported further waterway industrialization, including chemical, animal byproduct, fertilizer, and metal processing plants. Bulk marine shipments of coal, lumber, sand, gravel, and other building materials increased to support the population growth and associated residential development further inland from the industrial development along the creek (NYSDOT and FHWA 2005). Additional waterway development occurred in the early twentieth century, accompanied by further industrial development. Large barges, steamers, sailing vessels, and car floats used the waterway. Towing services also began circa 1900 to address shallow depths and bridge height constraints (NYSDOT and FHWA 2005). USACE-compiled traffic data indicate that there were at least 4,500 two-way vessel trips in the waterway circa 1890, before federally funded dredging extended above Vernon Avenue. Comparable numbers rose to more than 15,000 two-way vessel trips by 1900 and reached approximately 28,000 two-way vessel trips in 1917 (NYSDOT and FHWA 2005).

By 1910, Newtown Creek and its tributaries constituted the busiest waterway of its size in the world, with 65 to 70 industrial businesses receiving and shipping petroleum, chemicals, coal, and building products. Circa 1907, a 1,000-acre rail and marine terminal was constructed on the north side of Newtown Creek (east of Dutch Kills), with several short piers to handle heavy freight. Peak activity circa 1916 included 5.9 million short tons of cargo moved through the waterway. With vessel activity continuing to increase, the U.S. Congress approved a dredging project in 1921 to remove Mussel Island and create a Turning Basin. Mussel Island was a navigational hazard, and its removal aided the turning of larger vessels. It was removed circa 1932 (War Department 1931, 1932). Additional navigation improvements to the waterway occurred in the 1920s and 1930s, increasing raw materials shipped in and manufactured products shipped out. These improvements supported sugar refineries, hide tanning plants, canneries, copper wiring plants, and petroleum refineries (NYSDOT and FHWA 2005; see Appendix J). In 1935, according to USACE, 6,752 vessels passed the Kosciuszko Bridge (NYSDOT and FHWA 2005).

Between 1945 and 1965, waterfront use dropped by approximately 50%, due to an overall decline in manufacturing facilities and a shift to materials handling and improved highway access into the industrial area. Associated marine cargo shipments also declined during this time period. This decline has continued to the present, with annual marine cargo in and out of Newtown Creek dropping from 5 million tons in the 1960s to 1 to 1.5 million tons in the late 1990s and early 2000s (NYSDOT and FHWA 2005).

Since the early 1990s, marine traffic upstream of CM 0.5 has been primarily tug and barge traffic, with a small number of self-propelled coastal tankers (NYSDOT and FHWA 2005). Based on the Newtown Creek Navigation Analysis conducted in 2005, the most recent available, an average of one tugboat per day and one self-propelled oil tanker per week made inbound and outbound transits during a 14-year period between 1990 and 2003 (NYSDOT and FHWA 2005). The draft of these vessels was between 8 and 14 feet. More recently, the frequency of ship traffic was highest in the region between the creek mouth and the vicinity of Hugo Neu Schnitzer (aka SIMS Hugo Neu) (Hugo Neu) (DAR No. 125) docks, with lower frequency of ship traffic in the region upstream of Hugo Neu (see Section 5.3.2 of Appendix G).

As of 2005, there were 13 active marine facilities in operation along the waterway. This number includes Motiva Brooklyn Terminal (DAR No. 50), NYCON Supply Corporation (DAR No. 58), Hugo Neu (DAR No. 125; at approximately CM 1), Getty Terminals Corp. #58220 (Getty Terminals) (DAR No. 47), Metro Terminal (DAR No. 52), BP Products N America Brooklyn Terminal (DAR No. 48),[31] Empire Transit Mix, Inc. (DAR No. 59), Charles J. King, Inc. (DAR No. 206), Bayside Fuel Oil Depot – 1100 Grand St (Bayside Fuel

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[31] BP Products N America Brooklyn Terminal was sold to Kinder Morgan Operating L.P. "D," a Delaware limited partnership effective February 2, 2016. Nonetheless, to maintain consistency with the draft DAR, this facility is referred to as "BP Products N America Brooklyn Terminal."

Oil Depot) (DAR No. 51), Malu Properties/Former Ditmas Oil/Former Gulf Oil (DAR No. 123), Maspeth Concrete Loading Corp. (DAR No. 210), the Greenpoint Marine Transfer Station and Incinerator (DAR No. 11b), and Newtown Creek Water Pollution Control Plant, Brooklyn Wharf (no DAR No.; NYSDOT and FHWA 2005).

Commercial marine waterway use continues today with shipping service to various commercial businesses located along Newtown Creek, including materials recycling, bulk fuel transport, waste handling, and other industries. As of 2020, the following marine facilities are currently active along the waterway:

- Zenith Energy (formerly Brooklyn Terminal, DAR No. 50)
- Sims Metal (DAR No. 125)
- Allocco Recycling
- United Metro Energy (formerly Metro Terminal, DAR No. 52)
- Kinder Morgan (formerly BP Products North America Brooklyn Terminal, DAR No. 48)
- Empire Metal Trading (formerly Charles J. King, Inc., DAR No. 206)
- TNT Scrap
- Bayside Fuel Oil Depot – 1100 Grand Street (DAR No. 51)
- NYCDEP – Newtown Creek Water Pollution Control Plant

Shipping activity in Newtown Creek is expected to continue in the future.

3.2.4 Navigation Channel and Dredging History

A federally authorized navigation channel is present within the Newtown Creek main stem, including portions of the tributaries, as shown in Figure 3-14. This figure presents the authorized federal navigation channel depths for various portions of the creek as of 2009. The authorized depths have not changed from 2009 to the present. Actual channel depths are different than the authorized depths due to a lack of any maintenance dredging for many years, as described in this subsection. Dredging in Newtown Creek and its tributaries began in the mid-1800s. The Act Providing for Improvement of Newtown Creek, passed by the U.S. Congress on May 13, 1869, allocated funds for dredging Newtown Creek to a depth of 6 feet mean low water (MLW) across its entire width and length (BPL 1870). The Rivers and

Harbors Act (RHA) of 1884 provided appropriations for additional dredging in the creek. Two canals, present-day Whale Creek and the East Branch of Newtown Creek, were cut into the marshy edges of the waterway (NYSDOT and FHWA 2005). In 1890, English Kills was excavated by the City of Brooklyn under legislative authority of the State (War Department 1936c). A federally authorized navigation channel was established by the early 1900s to expand the main channel and dredge sediment from Dutch Kills, along with portions of English Kills, to create several slips (NYSDOT and FHWA 2005; War Department 1916). Further appropriations for dredging in the creek were a result of the RHA of March 2, 1919, and are directly quoted in the following list:

> a. A channel, 23 feet deep [relative to MLW], 130 feet wide, from the East River to 150 feet north of Maspeth Avenue
> b. A triangular area, 23 feet deep [relative to MLW], at the north side of the entrance
> c. A turning basin, 23 feet deep [relative to MLW], at Mussel Island
> d. A channel in the East Branch, 20 feet deep [relative to MLW] and 150 feet wide, and 2,000 feet long
> e. A channel in Dutch Kills, 20 feet deep [relative to MLW], 75 to 100 feet wide, where practicable, and 2,800 feet long; with a turning basin at the head
> f. A channel in English Kills, 20 feet deep [relative to MLW] and 150 feet wide to the Metropolitan Avenue Bridge, including the easing of bends; and thence 12 feet deep [relative to MLW], 100 feet wide, to within 80 feet of the Montrose Avenue Bridge, suitably widened at bends and materially widened by the excavation of upland mounds at the second bend above the Metropolitan Avenue Bridge (USACE 1973).

By September 1922, the Newtown Creek channel had been dredged to a width of 125 to 150 feet and a depth of 20 feet MLW from its mouth at the East River to Hobson Avenue, as proposed in the RHA of 1919 (War Department 1928). By 1929, Newtown Creek and Maspeth Creek were dredged to the depths proposed in the RHA of 1919 (War Department 1929a), but the proposed dredging of English Kills and East Branch (with depth modifications) did not occur until 1936 and 1945, respectively (War Department 1937, 1941, 1945). In 1930, there was a hearing before the War Department on the potential widening and deepening modification to the RHA of 1919. In this hearing, there were statements about a coal barge grounding above the Greenpoint Avenue Bridge and blocking an oil barge for a day and a half, the bottom of vessels being scraped when going up the creek due to shallow water, and damage to the bottom of barges due to insufficient water depth to reach bulkhead and barges having to be winch-dragged (War Department 1930a, 1936d). As a result of the lack of federal funding, the reduced drafts of vessels operating in Newtown Creek, and environmental concerns with impacted sediment, only 64% of the 1919 federally authorized navigation project has been completed (USACE 2015).

Sewerage was a byproduct of the intense nearby residential development that accompanied industrialization of the waterway. By World War I, the Borough of Brooklyn was dredging sewerage deposits. Siltation from sewers emptying into the creek became an acknowledged problem during the inter-war period, and the issue was primarily dealt with as a navigation difficulty (NYSDOT and FHWA 2005). Dredging to maintain the navigation channel in Newtown Creek and its tributaries continued through 1950, with the main channel (East River to Maspeth Creek) last dredged to the authorized depth of 23 feet MLLW in 1951 (USACE 2012).

Limited maintenance dredging activities continued through the mid-1970s, mostly in the tributaries (including areas to address docks and berths), but only extended to depths of 16 to 18 feet MLLW (USACE 2012). USACE dredging records from 2012 show that the last dredging activity was performed in 1974 to 16 feet MLLW in East Branch (USACE 2012). United States Code mandates appropriation of funds for dredging in federally authorized channels on a 5-year cycle (33 U.S. Code § 579a(b)(2)). No federal funding for navigational dredging has been allocated to Newtown Creek since 1974. Current vessel use and the lack of maintenance dredging supports the fact that current depths are adequate and removal of sediment to the authorized depths is not warranted to meet the current navigational needs of Newtown Creek.

In a 1985 Environmental Impact Statement for maintenance dredging of Newtown Creek and Dutch Kills, USACE wrote that "based on an analysis of the using traffic, it has been determined that dredging to less than the authorized depths would be satisfactory" because "the draft of most vessels using Newtown Creek are mostly 15 feet and shallower" (USACE 1985).

In 2014, navigational dredging was conducted in the first mile of Newtown Creek and in Whale Creek by NYCDEP to support access requirements for the residual waste management operations at the Newtown Creek WWTP. Approximately 30,000 cubic yards of material were mechanically removed to depths of approximately 17 to 19 feet MLLW (NYCDEP 2014a).

Various bathymetric surveys have been conducted in Newtown Creek between 1991 and 2019 (surveys completed by USACE in 2016, 2017, and 2019 were not used for purposes of the RI Report, as discussed in Section 3.1.3). The surveys were a mixture of full and partial spatial coverage throughout the creek, as well as varying between collecting single-beam and multi-beam bathymetry data. There are a number of conditions in the Study Area that create impediments to navigation, regardless of water depths, or that limit the degree to which dredging to authorized depths can be accomplished. These conditions include damaged infrastructure (e.g., broken lift bridges and railroad trestles); the presence of subsurface utility corridors at elevations above the authorized depths; shoaling that has resulted from deposition of CSO solids; or construction of low vehicle bridges that prevent access for many marine vessels.

3.2.5 Land Use and Zoning

Industrial development along Newtown Creek accelerated rapidly during the late 1800s. A tremendous growth in the population of nearby residential areas accompanied the commercial and industrial development (NYSDOT and FHWA 2005). In 1916, NYC adopted the Building Zoning Resolution. The resolution established three classes of use districts: residential, business, and unrestricted. Newtown Creek was classified as an unrestricted district, so any new facility engaged in trade, industry, and use was allowed to operate along Newtown Creek (NYC 1916).

Newtown Creek was an attractive location for manufacturers because it was close to international ports and population centers, but it offered more space and less zoning regulation than similar districts in NYC (MANY 1921; Brooklyn Chamber of Commerce 1923). A brochure published by the Brooklyn Chamber of Commerce in 1923 noted that dozens of acres ideal for heavy industry and subject to few restrictions located close to a plentiful supply of labor and transportation facilities for heavy freight were available in the Newtown Creek industrial district (Brooklyn Chamber of Commerce 1923). As additional manufacturing facilities and storage yards were built along the creek, more raw materials needed for manufacturing processes (e.g., ores, synthetic chemicals, oil, lumber, and coal) were available. Locating a business on Newtown Creek allowed manufacturers to minimize the costs associated with transporting both raw materials and finished goods (Hurley 1994).

In the 1920s and 1930s, the industrialized land bordering Newtown Creek and its tributaries, referred to as the Newtown Creek Industrial District, constituted a section approximately 3.5 miles long and from 0.25 to 0.75 mile wide; it included 80 to 85 active waterfront properties (MANY 1921; NYSDOT and FHWA 2005). In 1921, as shown in Figure 3-15, a large portion of the eastern end of the industrial district located near Maspeth Creek was still entirely undeveloped, with more than 550 acres of undeveloped land still existing directly on or near the creek (MANY 1921). By 1936, the Newtown Creek area was well served by highways and railroads and was extensively developed with commercial, industrial, and manufacturing uses (War Department 1936b).

By 1960, the 1916 zoning ordinance had been amended several times and included nine land use districts. Newtown Creek remained an unrestricted area (NYCDCP 1960). In 1961, NYC zoning ordinances were revised. Areas adjacent to Newtown Creek were designated as M3 Heavy Manufacturing Districts. These districts were characterized as "designed to accommodate the more essential heavy industrial uses, which involve more objectionable influences and hazards, and which, therefore cannot reasonably be expected to conform to those performance standards which are appropriate for most other types of industrial development" (NYCDCP 1961).

There have been few significant changes in land use near the creek since the 1960s. Pockets of mixed use, commercial, and sparse residential developments exist. However, the predominant land use in the Newtown Creek area remains industrial (see Figures 3-16 and 3-17; NYC 2011). The majority of the water frontage is developed for terminal or industrial purposes and is occupied by large-lot, truck-dependent, heavy industrial operations (NYSDOT and FHWA 2005; NCBOA 2012).

Although the broad use categories have not changed since the 1960s, additional regulations (e.g., overlays) have been incorporated into the zoning ordinances and apply to Newtown Creek and surrounding neighborhoods. Overlaying economic development designations for the Newtown Creek area include the East Williamsburg, North Brooklyn, and Maspeth Industrial Business Zones (NYC) and the Brooklyn Navy Yard/East Williamsburg Empire Zone program (New York State) (NCBOA 2012).

Industrial business zones were created in 2006 that established areas within NYC where expanded business services are available for industrial and manufacturing businesses. Three industrial business zones—Long Island City, Maspeth, and North Brooklyn—are located along Newtown Creek (NYCEDC 2016b). In 2011, New York City Department of City Planning conducted a survey to identify open industrial use (OIU) facilities operating within the industrial business zones. The study characterizes OIU facilities as “unenclosed businesses which unless managed properly can have negative impacts on air soil, and water quality.” The draft *Open Industrial Uses Study* was published in 2014 and identified 65 OIU facilities occupying approximately 37 acres along (or near) Newtown Creek. OIU facilities identified in the study include 28 unenclosed storage facilities; 24 automobile-dismantling facilities; five construction and demolition transfer facilities; five scrap metal processing facilities; two concrete and asphalt facilities; and one waste recycling facility (NYCDCP 2014).

Due to the prevalence of commercial/industrial land use in the vicinity of Newtown Creek, most of the ground surface around Newtown Creek is impervious to infiltration. Approximately 78% of the 10-square-mile area that potentially contributes to groundwater discharge to subsurface sediment in the Study Area is impervious (see Section 5.1.1 of Appendix F).

3.2.6 Historical Industrial Operations

By the late 1800s, Newtown Creek had become an integrated industrial district in which raw materials and finished products were often transferred from one facility to another (Hurley 1994). In 1884, the *Brooklyn Eagle* reported that “there are probably few, if any, places in the vicinity of New York where so many valuable industries are located on so small a space of water as are to be met with on Newtown Creek” (BPL 1884). Industrial operations located on and near Newtown Creek were documented in survey reports conducted by the War Department, articles published in newspapers (e.g., the *New York Times* and *Brooklyn Eagle*), and reports from public agencies (e.g., New York State Department of Health [NYSDOH] and Brooklyn Common Council). Given the variety and number of industries that have operated along the creek since the early 1800s, descriptions of all historical processes are not feasible in this report. However, detailed information about numerous historical industries is provided in the draft DAR (see Appendix J). The draft DAR provides a summary of the historical data collection and review process undertaken to date, consistent with the RI/FS Work Plan (AECOM 2011), Data Collection Plan (Anchor QEA 2011c), and RI/FS. The draft DAR reviews the collection and evaluation of historical documentation and data pertaining to the Study Area itself, as well as the collection and evaluation of pre-existing documents and data from upland areas adjacent to the Study Area. The upland area data collection activity is intended to help identify potential significant historical and ongoing sources of COPCs to the Study Area.

Brief summaries of some of the more prominent historical industries are presented in the following subsections. The objective of these summaries is to provide an overall description of the types of operations associated with each type of industry, including the variety of raw materials used, byproducts of the manufacturing processes, waste streams, contaminants, and finished products used and/or manufactured along the creek. Much of this material is derived from Section 4 of the draft DAR (see Appendix J). Section 5 of the draft DAR provides a summary for 150 sites of historical operations, potential COPCs, and potential transport pathways from these sites to the Study Area (e.g., overland, bank erosion, stormwater). Figure 5-1 and Tables 5-2 and 5-3 from the draft DAR provide information regarding the locations of DAR sites in relation to the Study Area, historical and current site use, historical and current potential areas of concern, potential COPCs, and transport pathways to the Study Area. Appendix B to the draft DAR includes upland site summaries for the non-respondent sites. A summary of the respondent sites is included in Section 5.6 of the draft DAR, and detailed summaries are included in Appendix C of the draft DAR. Section 6 of the draft DAR identifies a preliminary list of potentially significant sources from the list of 150 sites, including why they are considered potentially significant; Section 7 of the draft DAR provides additional detailed discussion for 11 of these potentially significant sites, with details on the transport pathways. Following the submittal of the draft DAR, more specific evaluations of potential ongoing source pathways (i.e., point sources and overland flow, groundwater, the East River, bank erosion, atmospheric deposition, overwater activities, and contaminant seeps) were conducted as part of the RI, and the results of those evaluations are summarized in Section 5 of this RI Report. In October 2018, nine new upland site summaries were developed, and six existing upland site summaries were modified; these are included as an addendum to Appendix J. These upland site summaries include information about the potential for these sites to represent significant ongoing sources to Newtown Creek. Controlling ongoing sources is a critical step in the management of contaminated sediment sites before remedies are undertaken. Clearly, many historical industries may have released, or did release, contaminants that are the focus of this RI (e.g., PAHs and other hydrocarbons, PCBs, metals, volatile organic compounds [VOCs], semivolatile organic compounds [SVOCs]) into Newtown Creek. The draft DAR and upland site summaries attempt to summarize contaminants and transport pathways to Newtown Creek from many of these historical operations, but it is not possible to quantitatively determine the magnitude nor the timing of potential releases throughout the Study Area that may have contributed to the nature and extent of contamination in Newtown Creek sediments. Finally, not every type of historical industrial operation is included in the following subsections (e.g., acid works, cordage works), nor is the level of detail the same for each industry, due to the availability of historical information about these industries.

A number of the DAR sites are discussed throughout this RI Report. Additional information about these sites can be found in the draft DAR (particularly Figure 5-1 and Tables 5-2 and 5-3; see Appendix J). DAR sites that are located on the shoreline directly adjacent to Newtown Creek are also shown in Figures 3-18a through 3-18j of this RI Report. Although these particular sites are located adjacent to Newtown Creek, because the current nature and extent of contamination in the creek is a function of many historical and current fate and transport processes as well as a variety of pathways, contamination proximate to these shoreline sites was not necessarily derived from these sites.

Sections 3.2.6.1 through 3.2.6.23 summarize the industrial operations and cross-reference the specific sections of the draft DAR where the detailed information can be found. Note that the amount of information provided in this section and in the draft DAR depends on the amount of information available at the time of writing, which varies depending on the type of industrial operation.

3.2.6.1 Adhesives Factories

Adhesives factories began operating around the creek in the early 1920s. Synthetic adhesives can be either thermoplastic or thermosetting. These were historically used in the manufacturing of screen-printing inks; tobacco products; shoemaking; pressure-sensitive tapes and labels; sealing devices; insulated wire; and other fabricated rubber products (USEPA 1976a). There are a number of types of adhesives: water- and solvent-borne adhesives, pressure-sensitive adhesives, hot-melt adhesives, and structural adhesives, but adhesives are typically a polymer resin (synthetic or natural) dissolved in a solvent (water or organic solvent). Water- and solvent-based adhesives are two of the largest classes of adhesives. Adhesives and sealants use natural, raw materials such as tar, resin, animal protein, and casein, as well as natural rubber and synthetic polymers.

3.2.6.1.1 Contaminants Associated with Adhesives Manufacturing

Synthetic solvent-based adhesives comprise polymers, including, but not limited to, polychloroprene, polyurethane, acrylic, silicone, and natural and synthetic rubbers (elastomers). Organic solvents contain VOCs, including, but not limited to, formaldehyde, toluene, xylene, and alcohols (such as methanol and ethanol), depending on whether adhesives are organic solvent-based or water-based.

3.2.6.1.2 Adhesives Manufacturing Sites near Newtown Creek

Some of the first adhesive factories were located just north of Newtown Creek in Long Island City, Queens, or just south of the creek in Greenpoint, Brooklyn (NYSDOH 1965). These include Manhattan Adhesives Corporation; Stein, Hall and Co. Adhesives; and several others (CCBQ 1922). Over the next several decades, the trend of erecting adhesive factories along the banks of Newtown Creek persisted. By 1980, Manhattan Adhesives Corporation; Polymer Industries, Inc.; Slomons Laboratories; Atlantic Paste and Glue; Gill Chemical Products, Inc.; United Resin Products; and Northern Adhesives, Inc., were located between Long Island City and Greenpoint (NYSDEC 1976; NYSID 1980).

During the 1960s, Manhattan Adhesives Corporation began operation at 425 Greenpoint Avenue, where Metro Terminal (DAR No. 52; NYSID 1963) is located (see Table 5-3 and Figure 5-1 of the draft DAR in Appendix J). In 1983, there was an investigation into whether 400,000 gallons of oil contaminated with PCBs were being illegally stored at the then-abandoned Manhattan Adhesives Plant (Golubski 1983). Today, the only remaining historical adhesives company located in the Newtown Creek vicinity is Northern Adhesives, Inc., located approximately 750 feet from the creek in Brooklyn.

3.2.6.2 Animal Rendering, Glue Factories, Fertilizer Plants, and Acid Works

Slaughterhouses, soap, candle/tallow, hide tanning, glue factories, fertilizer manufacturers, acid works, and other industries that utilized animal remains and timber were operating near Newtown Creek by the 1850s. Raw materials for these operations included dead horses and other animals, fish scraps, refuse material, green bones, wood, and plant biomass. The animal byproduct materials were collected from butchers' scraps and garbage, and were conveyed to the factories in covered wagons. Specific operations associated with animal rendering industries included bone boiling, bone burning, scrap drying, grease extraction, and bone grinding (USDOT and NYCEDC 2004; *Harper's Weekly* 1881; BPL 1871, 1884, 1890). Operations associated with acetic acid and sulfuric acid production included pyrolysis, calcination, or destructive distillation of wood or sulfurous feedstocks stills (*Scientific American* 1857; FAO 2019).

These operations did not always occur at the same facility. Smaller facilities often performed one operation in the process and sold the product to another facility for additional processing. For example, bone black produced by burning bones was purchased and used by the sugar refineries. The fumes from bone burning were captured and condensed to produce sulfate of ammonia, which was used in the manufacture of fertilizers (BPL 1893).

3.2.6.2.1 Contaminants Associated with Animal Rendering, Glue Factories, Fertilizer Plants, and Acid Works

Animal rendering operations produce large volumes of wastewater, which can impair water quality if discharged. Discharges from such operations would have included organic matter that would have a negative impact on the DO and total Kjeldahl nitrogen (the sum of organic nitrogen and ammonia) concentrations in surface water in the creek. Oil and grease, fecal coliform, extreme temperatures, and variations in pH would have also been likely components of these discharges.

In the eighteenth and early nineteenth centuries, the largest nuisance of fertilizer manufacturing was considered to be the odor it produced. Fertilizer manufacturers used residual sludge from the petroleum refining process (BPL 1886; Baker and Kent 1887). Sludge acid was a black liquid having the consistency of molasses. To manufacture fertilizer, the sludge acid was mixed with ground animal bones and water, producing “a most abominable stench” (NYT 1881a). To produce a more tolerable odor, sludge acid was sometimes replaced with sulfuric acid. Although some facilities purchased sludge acid from petroleum refineries, others obtained sulfuric acid from chemical manufacturers. At the Read Fertilizer Co., sulfuric acid was pumped from the tanks at the manufacturer Nichols & Co. directly into the upper stories of the fertilizer factory (*Harper’s Weekly* 1881; BPL 1893).

The process used by chemical manufacturers to produce acetic acid involved pyrolysis of wood, which produced charcoal and tar as a byproduct. Typical pyrolysis yields per ton of air-dried wood from northern deciduous hardwoods are approximately 50 kilograms (kg) acetic acid; 50 kg insoluble tar as a waste product, which could be used as a wood preservative; and road tar (FAO 2019). Sulfuric acid was historically produced by the heating of sulfur-rich feedstocks such as blue or green vitriol, and byproducts included sulfurous gases when incomplete oxidation of feedstocks occurred in the Pb chambers (Mousavi 2012; Otterson 1886). Sulfuric acid was used in fertilizer production and petroleum refining (Kiefer 2001).

Ground animal bones were also used for the manufacture of bone black, a natural black pigment created by the destructive distillation of animal bones in the absence of oxygen, (Ebonex Corporation 2019; NYSBOH 1896). A by-product of this pyrolytic process is a liquid consisting of tar (Dippel’s oil or bone oil) and pyrolytic water (Purevsuren et al. 2004, 2017). The oil—which would have consisted mostly of aliphatic chains, with nitrogen functionalities, and which includes species such as pyrroles, pyridines, and nitriles, as well as other nitrogenous compounds—would have been dumped in pits on site and, with time, could have impacted Newtown Creek.

3.2.6.2.2 Animal Rendering, Glue Factories, Fertilizer Plants, and Acid Works Sites near Newtown Creek

Glue manufacturing began on Newtown Creek circa 1840, with Peter Cooper's Glue Manufacturing plant, where the business of glue making and gelatin making was conducted until 1915 (*Brooklyn Eagle* 1891; BPL 2011). The 10-acre factory, consisting of nearly 30 buildings, was located on Maspeth Avenue, where the current National Grid Greenpoint Energy Center (DAR No. 32) resides (see Section 5.6.4 of the draft DAR in Appendix J for more information on DAR No. 32). Smaller glue manufacturing facilities existed, but focused primarily on fertilizer operations; one such facility was Preston Fertilizer Company, which also rendered bones for glue products. The company operated on Lot 279 of the Review Avenue Development I (RAD I) (DAR No. 41) site from 1884 to 1899, when it became the American Agricultural Chemical Company and operated until 1915 (NYT 1884, 1899; NYSBOH 1889). See Table 5-3 and Figure 5-1 of the draft DAR in Appendix J for further details on these DAR sites, including the description for Malu Properties/Former Ditmas Oil/Former Gulf Oil (DAR No. 123) (previously known as Former Gulf Oil Corporation – Greenpoint Bulk Plant), which lists animal rendering, bone boiling, and glue manufacturing as historical site uses. Also see the updated upland site summary for RAD I (DAR No. 41) in Appendix J.

As part of the original manufacturing of glue, a process of boiling animal bones and other connective tissue, wastewater rich in lime would be produced. Drainage from the liming tanks would pass through a paved basin in the Peter Cooper yard and then directly discharge to Newtown Creek (NYSBOH 1894). In addition to the wastewater discharges to the creek, an offensive odor would be produced from the boiling process.

Fertilizer plants were closely linked with animal rendering, and several records exist regarding past practices for these operations. The Coe Fertilizer Company, located in Greenpoint Brooklyn, had one of the more offensive fertilizer plants, due to its odor and the disposal of garbage on the marsh flats. The plant was later relocated to Barren Island in 1877 (BPL 1877). In 1969, Ecology Inc., developed a refuse-to-fertilizer facility along the shores of English Kills, partly as a way of solving the growing garbage disposal problem, as available land for waste disposal was running out (NYT 1969). The property is now operated by Waste Management and no longer produces fertilizer from refuse (see Appendix J).

A number of acid works near Newtown Creek produced and stored acetic and sulfuric acid. An 1886 Department of Health Annual Report described the acid works of Bushwick Chemical Works, a predecessor of Honeywell, Inc., which operated near the English Kills tributary of Newtown Creek from approximately 1851 to 1915. This facility produced various acids including sulfuric, muriatic, nitric, and acetic acid with acid tar as a waste stream. Sulfurous gases were produced when incomplete oxidation of feedstocks occurred in the Pb chambers, and tar was formed as a waste product of acetic acid manufacture (Otterson 1886). General Chemical (also a predecessor to Honeywell, Inc.) in Queens manufactured sulfuric and acetic acid (NYSBOH 1896).

Although specific practices changed over the years, animal rendering, fertilizer, and soap manufacturing facilities, including the Long Island Soap Company, Joseph Rosenberg and Sons (later known as Rencoa, Inc. [DAR No. 139]), and Van Iderstine Company (later known as RAD I [DAR No. 41]), continued to operate along the creek until the 1960s and 1970s (War Department 1891, 1915, 1936a; Hazen and Sawyer 1960; see Section 7.6 and Table 5-3 of the draft DAR and upland site summaries in Appendix J).

3.2.6.3 Asphalt Production and Storage Operations

The production of asphalt for paving material by asphalt plants, as well as the storage and transfer of asphalt, are well documented in the vicinity of Newtown Creek from at least 1907 to the present day (Anchor QEA 2012n). Asphalt operations near Newtown Creek were documented in 1907, 1921, 1936, 1940, and 1976 in Sanborn maps, survey reports conducted by the War Department, and articles published in the *New York Times* and *Brooklyn Eagle*.

3.2.6.3.1 Contaminants Associated with Asphalt Production and Storage Operations

Pollutants generated from asphalt production, storage, and transfer are associated with raw materials such as petroleum and include TPH, VOCs, chlorinated volatile organic compounds (CVOCs), SVOCs, PAHs, metals, and particulates, as well as other asphalt modifiers and additives. Specific pollutants and contamination pathways vary, depending on which types of asphalts are being processed and what type of process is being used (CDC 2000) (see Table 5-3 of the draft DAR in Appendix J for Manhattan Poly Bag [DAR No. 130], Morgan Oil

Terminal, Brooklyn [Morgan Oil] [DAR No. 60], and RAD I [DAR No. 41]). These operations typically used aboveground storage tanks (ASTs) and underground storage tanks (USTs) to store oil, as well as conveyance piping, loading racks, and ancillary equipment used in oil storage and distribution associated with asphalt production. Some operations (e.g., Manhattan Poly Bag [DAR No. 130]) also used asphalt and tar tanks in asphalt production.

3.2.6.3.2 Asphalt Production and Storage Sites near Newtown Creek

Green Asphalt Company, LLC, a transfer station that processes asphalt waste, currently occupies a portion of RAD I (DAR No. 41) adjacent to Newtown Creek (see Sections 6.2.4 and 7.6 of the draft DAR and the relevant updated upland site summary in Appendix J). Several other asphalt production and storage sites have operated in the vicinity of Newtown Creek, specifically English Kills, including at the following DAR sites: Morgan Oil (DAR No. 60; see Sections 6.2.7 and 7.10 of the draft DAR and the relevant updated upland site summary in Appendix J); Manhattan Poly Bag (DAR No. 130; see Section 6.2.7 of the draft DAR and the relevant updated upland site summary in Appendix J); Perez Interboro Asphalt (DAR No. 202); Waste Management of NY – 161 Varick Avenue (DAR No. 215); and Meyer Fine Lumber Co. (DAR No. 218; see Table 5-3 and Figure 5-1 of the draft DAR in Appendix J).

3.2.6.4 Automobile Component Manufacture, Repair, and Service

By the early 1920s, several automobile manufacturers (automobile components, not assembly), service stations, garages, repair shops, and auto parts stores were already operating in the vicinity of Newtown Creek (CCBQ 1920).

3.2.6.4.1 Contaminants Associated with Automobile Component Manufacture, Repair, and Service Industries

Automobile component manufacturing is a complex production process that requires large amounts of material inputs and generates large amounts of waste. The manufacturing process includes foundry operations, metal fabricating, metal finishing, and painting/coating. Wastes resulting from the various stages of production range from air emissions from foundry operations to spent solvents from surface painting and finishing (USEPA 1995a). USEPA (1995a) generally characterizes these waste products as acid/alkaline wastes, cyanide wastes, waste oils, solvent wastes, and metal wastes. Specifically, these waste products can contain

SVOCs (including PAHs); VOCs; PCBs; TPH (which can include PAHs and C19-C36); solvent wastes, including 1,1,1-trichloroethane, acetone, xylene, and toluene; and metals, including Cu, chromium (Cr), and nickel (Ni). USEPA (1995a) details many of the wastes. Auto repair and service shops may generate many of the same waste products as part of their operations, and many of these repair and service locations also have on-site diesel, gasoline, lube oil, waste oil, solvent, and chemical storage tanks (see Appendix J).

3.2.6.4.2 Automobile Component Manufacture, Repair, and Service Sites near Newtown Creek

Ford, Pierce-Arrow, White Motor, and American LaFrance were among the most prominent automobile component manufacturers established in Long Island City[32] in the twentieth century (Goodwin and Associates 2012). Historical records indicate that auto repair shops were also established near Newtown Creek as early as 1906, and bus depots were present as early as the 1950s, several of which are still present today. A number of automobile impound lots currently operate within the vicinity of Newtown Creek including, but not limited to, two operated by the NYC Marshals, at the Malu Properties/Former Ditmas Oil/Former Gulf Oil (DAR No. 123; see Section 5.6.8 of the draft DAR in Appendix J) and B.C.F. Oil Refining, Inc. (B.C.F. Oil) (DAR No. 27; see Section 7.8 of the draft DAR in Appendix J), in Brooklyn. Auto repair shops have also historically operated at sites within the vicinity of Newtown Creek; for example, NYCT – Fresh Pond Depot (DAR No. 18), Queens West (Hunter's Point) Center Boulevard (DAR No. 133), and United Envelope (DAR No. 17) in Brooklyn and Queens. Spills, leaks, and general operations may have discharged the products and contaminants listed in Section 3.2.6.4.1 from the manufacture of automobile components, as well as auto repair and service locations, directly or indirectly to the Study Area via a number of transport pathways (see Section 3.2.11, as well as Table 5-3 and Figure 5-1 of the draft DAR in Appendix J).

3.2.6.5 Coal Processing, Handling, Storage, and Fuel Use

A kerosene (refined from coal) refinery began operating in 1854 (*Brooklyn Eagle* 1859; Beaton 1955), when the North American Kerosene and Gas-Light Company began operating

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[32] The communities of Blissville, Hunter's Point, Dutch Kills, Ravenswood, Astoria, and Bowery Bay merged in 1870 to create the new jurisdiction of Long Island City (Goodwin and Associates 2012).

a factory along Newtown Creek in Long Island City. Its operations were short-lived as the company went bankrupt in 1860 (*Morning Courier and New-York Enquirer* 1860). The discovery of crude oil in 1859 led to an increased demand for petroleum lamp oil and by 1861, every coal-oil refinery in the United States either switched to petroleum or went out of business (Petroleum Publishing, Inc. 1959). Between 1860 and 1950, coal yards were a prominent industry, typically processing raw materials and producing finished products (Goodwin and Associates 2012). As such, the handling, transfer, and burning of coal would have been an integral part of the industrial setting.

3.2.6.5.1 Contaminants Associated with Coal Processing, Handling, Storage, and Fuel Use

Numerous studies have documented the levels of contaminants in coal and the potential effects on aquatic organisms associated with the coal contaminants in the environment as a result of leaching from storage piles and accidental spillage (USEPA 1978; Ahrens and Morrisey 2005; Stout and Emsbo-Mattingly 2008). PAHs and trace metals found in the Study Area are contaminants known to be associated with the storage, handling, transfer, and burning of coal (see Table 5-3 and Figure 5-1 of the draft DAR in Appendix J for additional information on sites where these releases may have occurred or did occur historically (e.g., Bayside Fuel Oil Depot [DAR No. 51]; Morgan Oil [DAR No. 60]; and Empire Transit Mix, Inc. [DAR No 59]). A discussion of byproducts from the carbonization of coal (e.g., coal tar) is provided in Section 3.2.6.10.

3.2.6.5.2 Coal Processing, Handling, Storage, and Fuel Use Sites near Newtown Creek

Coal yards were located throughout the Newtown Creek area including, but not limited to, the sites currently known as the following: RAD I (DAR No. 41) (circa 1884); Empire Transit Mix, Inc. (DAR No. 59) (circa 1933) (Sanborn 1933, 1970; NY Telephone Company 1969, 1978); Morgan Oil (DAR No. 60) (1907 to 1933); and Bayside Fuel Oil Depot (DAR No. 51) (1910s to 1930s, 1965 to 1996). See Table 5-3 and Figure 5-1, and Sections 7.6, 7.9, 7.10, and 7.11, respectively, of the draft DAR for more information on these DAR sites, as well as additional upland site summary information for Bayside Fuel Oil Depot and Morgan Oil in Appendix J. In the early 1850s, the North American Kerosene Gas Lamp Company, which operated on the former Pratt Oil Works (POW) site in Queens (Waste Management of NY/Steel Equities [formerly POW] [DAR No. 56]) (Section 5.6.7 of the draft DAR in Appendix J), and the footprint of Quanta Resources a/k/a Review Ave. Development II (Quanta/RAD II) (DAR No. 39) and RAD I (DAR No. 41) (see Figure 5-1 of the draft DAR in Appendix J), would have stored and used coal in the production of kerosene most likely from asphalt using the Gesner process.

In the 1930s, along the Queens side of Newtown Creek near the East River (known as Hunter's Point), as much as 600 tons of coal were unloaded from barges every day (NSRCNJ 1930). By 1950, fleets of barges, tugs, tankers, and "steam colliers" transported an estimated 7.7 million tons of materials, including coal, throughout the creek (Turner 1951; Goodwin and Associates 2012). The Morgan Oil site had two mooring areas for barge access and would load and unload petroleum products and coal from barges (EDR 2010; see Section 7.10 of the draft DAR in Appendix J).

English Kills was the site of numerous coal supply companies, such as the Buffalo Coal Yard, Knapp Coal Co., Burns Brothers Coal Co., and Scranton and Lehigh Coal Co. (Sanborn 1933; Goodwin and Associates 2012).

Coal was also delivered to and stored at a number of industrial sites for power production, including, but not limited to, sugar refineries located at the mouth of Newtown Creek, including the Havermeyer Sugar Refining Co. located at the NYCT Crosstown Annex Facility (DAR No. 20) circa 1887 and the National Sugar Refining Co. located at the Anheuser Busch Distributors of NY (DAR No. 208) site circa 1915 to 1970 (Sanborn 1915, 1970; see Table 5-3 and Figure 5-1 of the draft DAR in Appendix J).

3.2.6.6 Creosote Production and Treatment

Creosote production was ongoing in the Newtown Creek area as early as 1878. Eppinger & Russell Co. built one of the world's largest creosoting works on Newtown Creek and operated it for 70 years from 1878 through at least 1949 (ASME 1892). In 1891, a creosote oil tank explosion occurred at the Eppinger & Russell Co. plant located at Queens Block 65, Lot 57, which burned the Long Island City manufacturing plant to the ground on

December 10 of that year. The resulting fires spread a mile up Newtown Creek and produced damages equaling more than $40,000 (BPL 1891). Creosote production began to rise dramatically a few years later when Oscar Bernuth, chairman of Eppinger & Russell Co., introduced the Rueping process of wood preservation to the United States in 1903 (NYT 1958). The demand for creosote production rose again in 1918, when Jess Eppinger pioneered the practice of charring, coating, and impregnating timber with creosote to strengthen and artificially preserve the wood, as well as to deter boring insects from penetrating the material (WSJ 1943).

3.2.6.6.1 Contaminants Associated with Creosote Production and Treatment

Creosote is formed as a byproduct of coal tar distillation (coal tar creosote) or the thermo-decomposition of wood or fossil fuel (wood tar creosote). Depending on the process and resulting derivatives, creosote contains a mixture of hazardous chemicals, including carbolic acid, PAHs, cresols, and other phenols (USDHHS 2002).

3.2.6.6.2 Creosote Production and Treatment Sites near Newtown Creek

In addition to the Eppinger & Russell Co. site discussed in Section 3.2.6.6, there are instances of contamination attributed to creosote at sites including, but not limited to, Queens West (Hunter's Point) Center Boulevard (DAR No. 133) and Outlet City (DAR No. 132) in Long Island (see Table 5-3 and Figure 5-1 of the draft DAR in Appendix J). West Disinfecting Company located at Outlet City stored creosote for the manufacturing of the disinfectant "Coroneleum" circa 1901 to 1962 (see Table 5-3 of the draft DAR in Appendix J). From at least 1924 to 1952, creosoted planking was stored by the City of New York beneath the surface of Whale Creek (NYT 1952). In addition, creosote is most commonly used for treating timber, utility poles, and rail ties, so timber bulkheads, docks, and pilings may represent current or historical sources of creosote throughout Newtown Creek.

3.2.6.7 Distilleries

Several distilleries operated throughout history in the vicinity of Newtown Creek. The earliest cited distillery, Bache, Sons & Co. rectifying distillery, was erected in 1811 in the Fulton Ferry neighborhood of Brooklyn, and the industry expanded from there (American Publishing 1890). *Rectifying* is any process or procedure whereby distilled spirits are cut, blended, mixed, or infused with any ingredient that reacts with the constituents of the distilled spirits and changes the character and nature or standards of identity of the distilled spirits. *Blending* is the term commonly used today. By 1850, six distilleries, three rectifying establishments, and a brewery were present along the banks of the East River and Newtown Creek. Grain, corn, rye, oats, barley, and wheat were among the most common materials distilled, and machinery was primarily run using coal, as in most industries during that period. During these years, the typical manufacturer consumed up to 600,000 bushels of grain and 2,340 tons of coal to produce more than 950,000 gallons of whiskey, gin, and cologne per year (American Publishing 1890). By 1867, the Brooklyn-Queens distillery industry, then concentrated on the 3-mile stretch of Newtown Creek between Greenpoint and Hunter's Point, had grown even larger than what existed in 1850.

3.2.6.7.1 Contaminants Associated with Distilleries

The stench associated with the byproducts of distilling, including sour mash, swill, and waste, led residents to call for an investigation by the New York State Board of Health in 1880 (NYSBOH 1884). Discharges likely were high in organic matter and increased biochemical oxygen demand loadings in Newtown Creek. Coal-storage contamination is discussed in detail in Section 3.2.6.5.

3.2.6.7.2 Distillery Sites near Newtown Creek

Columbus Distillery, Blissville Distillery, Eastern Distillery Company, and Gaff and Fleischmann's Yeast Factory and Distillery were some of the largest distilleries on the east coast (BPL 1868, 1916; NYT 1881b). A number of these distilleries were located in Blissville, south of Sunnyside, between Newtown Creek and what is currently the Long Island Expressway.

3.2.6.8 Electronics and Electroplating Industries

The electronics industry includes the manufacture of passive components, semiconductor components, printed circuit boards, and printed wiring assemblies (World Bank Group 1998). Electroplating is the process of building up a layer of one metal on another (ILO 1972). Electroplating operations are documented in the vicinity of the creek as early as

1940 and also continue to operate in the area to the present day (see Table 5-3 and Figure 5-1 of the draft DAR in Appendix J).

3.2.6.8.1 Contaminants Associated with Electronics and Electroplating Industries

The electronics and electroplating industries produce a variety of solid, liquid, and hazardous wastes, as well as air emissions. The electronics industry's primary wastes are organic solvents and acids (USEPA 1995b). The electroplating industry uses acid/alkaline solutions, heavy metal-bearing solutions, and cyanide-bearing solutions in the production process (USEPA 1995c). Waste streams and/or releases to Newtown Creek would likely have contained cyanide, PCBs, SVOCs (including PAHs), VOCs, TPH, and heavy metals and are associated with spills, leaks, and general operations of the electronics and electroplating industries near the Study Area.

3.2.6.8.2 Electronics and Electroplating Sites near Newtown Creek

Historical electroplating operations occurred in Maspeth at Electronic Plating Corp. (DAR No. 112), also known as S&L Metal Products Corp. (DAR No. 142), from circa 1979 to 2001; in Brooklyn at Hardchrome Electro Plating, Inc. (DAR No. 124), from circa 1983 to 2005, Architectural Coatings, Inc. (DAR No. 103), from 2002 to 2011, and Technical Metal Finishers (DAR No. 43), from 1940 to 1982; and in Queens at Berger Industries (DAR No. 105), from circa 1972 to 1995. Present-day electroplating occurs at Wilco Finishing Corp. (DAR No. 149) in Brooklyn and Structural Processing Corp. (DAR No. 144) in Queens, with operations dating back to 1985 and 1986, respectively (see Table 5-3 and Figure 5-1 of the draft DAR in Appendix J).

3.2.6.9 Incinerators

Historical refuse incinerators (also known as destructors or municipal waste combustors) operated in the area from the early 1900s to the 1990s. Generally, historical incinerator operations involved receipt of refuse at the facility (transported by wagon, truck, or barge), sorting, shredding, incinerating, quenching of burning residue, and disposing of ash (Municipal Engineers of New York City 1951; Walsh et al. 2001). Ash was often sent to landfills or sold for construction-related uses, including the manufacture of cinder blocks or for land creation/filling uses (Zarin 1987; NYCDOS 1953; see Section 4.1.2 of the draft DAR in Appendix J).

3.2.6.9.1 Contaminants Associated with Incinerators

As described in Section 3.2.6.9.2, a number of historical incinerators were located in the area around Newtown Creek. There was a waste dump associated with at least one of the early incinerators adjacent to Newtown Creek at Apollo Street in Brooklyn. Air discharges, ash, fuels for the incinerator units, and solid waste associated with operation of the incinerators likely caused some contaminants found in municipal waste and incineration waste streams to enter Newtown Creek. These include, but are not limited to, metals, PCBs, dioxins, and PAHs (Dyke 2002; USEPA 1987; Guo et al. 2014; Liu et al. 2013). There are several pathways by which ash with contaminants from the municipal incinerators could have reached Newtown Creek, including aerial deposition, spills from barges, and overland runoff. Of note, PCBs would have been found in a variety of municipal solid wastes that would have been disposed of in the incinerators after their widespread commercial use began in the 1930s. These include, among other things, municipal trash containing the following:

- Inks in which PCBs were used
- Paints in which PCBs were used
- Adhesives in which PCBs were used
- Carbonless copy paper in which PCBs were used (starting in the late 1950s and 1960s)
- Plasticizers in which PCBs were used

Incineration of municipal trash with PCBs was particularly problematic because of the stability of PCBs at high temperatures (Carotti and Smith 1969). PCBs that were not combusted would have attached to ash and may have reached the creek.

Ash and air emissions from incinerators can also contain cadmium (Cd), Pb, mercury (Hg), other metals, dioxins/furans, PAHs, and PCBs (National Research Council 2000). The quantities of these chemicals potentially released to Newtown Creek are unknown.

3.2.6.9.2 Incinerator Sites near Newtown Creek

The first incinerator directly on the creek was constructed in 1929, behind the waterfront dump, which was located at the foot of Apollo Street and Norman Avenue (NYT 1931). Initially operated by the Brooklyn Ash Removal Company under contract with NYC, the City of New York took over the operation of the plant in 1934 (NYT 1934). Serious problems were associated with its operations, as evidenced by complaints lodged in the early 1950s by representatives of the Oil Workers Union, whose members worked at refineries located near the incinerator. At the dump and incinerator site, garbage and ash were dumped from trucks into scows and barges moored on the creek. An electric bucket conveyor used to load ash into barges at a rate of 35 cubic yards per hour was also present on the dock (USACE 1953). The union reported that the height of the dock, which was contiguous to the incinerator, was sufficient to cause vast and continuous billows of dust, soot, and ash clouds to rise, blow, and scatter over the area in proximity to the dock. Hundreds of nearby workers wore goggles and dust masks, but despite these precautions, the conditions were such that the union claimed that the workers' health was "endangered, impaired and broken." Fires on the scows and barges were apparently common, "occurring at all times of the day and night" (OWIU 1951). According to the union, "vast clouds of dust" emanated from the unscreened and uncovered dumping and handling of garbage and ash at the facility.

In 1959, New York City Department of Sanitation (NYCDOS) replaced the facility on Apollo Street with a new facility located at Kingsland Avenue in Brooklyn, adjacent to Newtown Creek, which continued to operate until 1994 (NYCDOS 1956, 1959). Although the Apollo Street and Kingsland Avenue facilities are often referred to as the "Greenpoint incinerator" in historical documentation, they were actually two separate facilities. As early as 1974, environmental compliance issues related to air emissions (potentially with Cd, Pb, Hg, other metals, dioxins/furans, PAHs, and PCBs) and effectiveness of pollution controls plagued the Greenpoint incinerator until its closure in 1994 (NYSDEC 1974a, 1990; Dvorkin 1977; Gilberg 1982; Work on Waste 1994).

NYCDOS opened the Betts Avenue incinerator, located at 58-73 53rd Avenue in Queens, in 1926, and it ceased operations in 1938. Following extensive renovations, NYCDOS reopened the Betts Avenue incinerator in 1950, and it operated until 1993. The Maspeth incinerator (which was the first constructed in the Study Area), located at Flushing and Metropolitan

Avenue in Queens and owned by NYCDOS, opened in 1916 and was closed in 1937 (NYCDOS 1930; Walsh et al. 2001).

3.2.6.10 Manufactured Gas Plants[33]

MGPs produced gas from coal or other organic materials. In the first half of the 1800s, MGPs primarily used bituminous coal to produce illuminating gas for use in municipal lighting (Moore Binder 1955; NYSDEC 2012a). By the late 1800s, manufactured gas was a widely used fuel for heating and cooking. By the 1950s and 1960s, as electricity and natural gas became available to more consumers, the majority of MGPs ceased production (NYSDEC 2012a; Murphy et al. 2005; Hamper 2006). In general terms, manufactured gas operations involved production of gas, purification of gas, handling of tars and oil byproducts, and handling of emulsions when tars and oils did not fully separate. Byproducts of the gasification process included coal tar, light oil, coke, and ammonia liquor. In some cases, these materials were sold for use by other industries (Murphy et al. 2005). Specifically, tar and tar-like material associated with the manufactured gas process have historically been used, and are currently used, by dozens of major industries, such as in ship building; production and use of creosote for preservation of pilings and timbers; steel manufacture; aluminum smelting; the production of dyes; the waterproofing of rope; tanning agents; manufacture of coal tar epoxy; the building industry (e.g., roof waterproofing, foundation sealants); asphalt sealants; and hundreds of other uses (USDHHS 2014).

3.2.6.10.1 Contaminants Associated with Manufactured Gas Plants

The primary contaminants associated with the tar byproduct of the manufactured gas production include PAHs and benzene, toluene, ethylbenzene, and xylene (BTEX) compounds. Other contaminants associated with the manufactured gas production process also include metals, SVOCs, and TPH compounds. Releases of these contaminants to Newtown Creek may have occurred via numerous transport pathways. Coal-storage contamination is discussed in detail in Section 3.2.6.5 (NYSDEC 2012a).

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[33] National Grid prepared the historical industry summary for its sites in Section 3.2.6.10 and subsections, and the industry information and site status summary are according to that company.

3.2.6.10.2 Manufactured Gas Plant Sites near Newtown Creek

The Brooklyn Union Gas Company operated a gas plant (Greenpoint Energy Center [DAR No. 32]) from 1927 until 1952, when the company switched to supplying natural gas (Goodwin and Associates 2012) (see Section 5.6.4, Table 5-3, Figure 5-1, and Appendix C [specifically, the relevant Respondent site summary] of the draft DAR in Appendix J of this RI Report). From the late 1960s through the present day, liquid natural gas was stored at the site. Substitute natural gas production also occurred at the site from the mid-1970s until 1985. Brooklyn Union Gas also operated a second, smaller MGP, Equity Works (DAR No. 33), near Maspeth Avenue and Vandervoot Avenue between 1903 and 1933 (see Section 5.6.5, Table 5-3, Figure 5-1, and Appendix C [specifically, the relevant Respondent site summary] of the draft DAR in Appendix J of this RI Report).

A remote gas distribution holder, the Scholes Street Holder Station, which was located approximately 750 feet from (but not adjacent to) the upper end of English Kills, was constructed in 1892 and potentially operated until 1951, when the gas holder was decommissioned. Gas was not manufactured at the site, there was no conveyance to English Kills, and there is no groundwater plume or migration pathway to English Kills (Arcadis 2014).

3.2.6.11 Metal Production, Smelting, and Metalworks and Fabricating[34]

Copper smelting and refining within the Newtown Creek industrial area commenced in the late 1800s. Feedstock consisted mainly of ore concentrates. The copper smelting process involved high temperature treatment of the copper concentrate in furnaces and converter vessels to separate the copper from impurities. The copper from the smelter was further purified by the refining process, which used fire refining in furnaces or electrolytic refining in tanks of copper sulfide and sulfuric acid. The resulting pure copper was then cast or shaped as desired. Used solutions from the electrolytic refining process were used to produce commercial products, such as copper sulfate (Craven 2000). Other metal-working and fabricating industries, including aluminum production, operated throughout the 1900s.

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[34] PDRC prepared the historical industry summary for its site in Section 3.2.6.11 and subsections, and the industry information and site status summary are according to that company.

3.2.6.11.1 Contaminants Associated with Metal Production, Smelting, and Metalworking and Metal Fabricating Industries

The most common contaminants associated with the metals smelting and processing industries are metals. However, similar to other heavy industries that historically operated on Newtown Creek, the metals smelting and processing industries also would have utilized fuels, solvents, and lubricating and hydraulic oils. As such, BTEX, TPH, PAHs, and SVOCs also would have been common contaminants associated with these industries.

3.2.6.11.2 Metal Production, Smelting, Metalworks, and Fabricating Shops near Newtown Creek

The PDRC Former Laurel Hill Site (DAR No. 16) (see Section 5.6.1, Table 5-3, Figure 5-1, and Appendix C [specifically, the relevant Respondent site summary] of the draft DAR in Appendix J of this RI Report), located near the confluence of Maspeth Creek and Newtown Creek, was used for the manufacture and distribution of acids beginning in 1870 and eventually expanded to include copper smelting, Bessemer copper converting, copper refinery operations, and other chemical and metals processing and production (Anchor Environmental 2007; NYSDEC 2003; Sanborn 1902). Smelting operations at this site were discontinued in the 1960s; electrolytic refining and other remaining operations were discontinued by 1984. Under an Order on Consent with NYSDEC, site remediation has been performed, including the following: 1) demolition in 1999 to 2000; 2) soil removal in 1987 and 2004; 3) groundwater barrier wall and groundwater collection/treatment system in 2005 and 2006; and 4) construction of caps on several site parcels completed as of 2018 (see Section 5.6.1 and Appendix C [specifically, the relevant Respondent site summary] of the draft DAR in Appendix J of this RI Report). Most of the site has been sold and is being redeveloped. PDRC currently has an ownership interest through a joint venture or direct ownership in approximately 4.8 acres of the original 37 acres.

A 90-acre aluminum production plant was constructed in February 1943 at the junction of the East Branch of Newtown Creek and Maspeth Creek. The plant was designed to produce an estimated output of 130,000 tons per year of aluminum. The plant utilized the typical technology of the day to produce aluminum from bauxite ores, using an electrolytic process in eight potlines housed in 16 buildings on the site. The plant also produced various alloys of aluminum and other metals during its tenure. The plant was placed on standby in May or June 1944 and was then dismantled for use as a U.S. Navy yard around September 1944 (K&L Gates 2014; Wilson 1944).

Other metals facilities along the creek included James B Davis Foundry (circa 1909), JR Elkins Inc. (circa 1950 to 1970), and Stoll Metal Corp. (circa 1947).

3.2.6.12 Metal Scrap and Storage

In the 1950s, with water traffic increasingly dominated by tugs and barges, businesses specializing in handling barge loads of scrap metal and building materials became proportionately more significant users of the creek (NYSDOT and FHWA 2005; Goodwin and Associates 2012). Common sources of recycled metals historically and currently processed by these facilities include automobiles and automotive scraps; transformers; appliances; steel drums; boat scrap; railroad and railcar scrap; mill scrap; and used construction beams, plates, pipes, tubes, wiring, and shot (OSHA 2008).

3.2.6.12.1 Contaminants Associated with Metal Scrap and Storage Industries

Scrap metal processing facilities receive bulk quantities of a wide range of scrap metal that historically was not sorted or screened for fluids prior to processing. Scrap processed at these facilities oftentimes was contaminated with oils, greases, lubricants, paints, and petroleum waste products from machinery, which contained VOCs, SVOCs, PAHs, and metals, among other chemicals. PCBs are a common contaminant associated with recycled metals, since the beginning of their commercial use in 1929. PCBs were present in electrical equipment, automobiles, refrigerators, washing machines, and televisions, as well as many oils, paints, sealants, and lubricants found on scrap metal.

3.2.6.12.2 Metal Scrap and Storage Sites near Newtown Creek

One of the sites that processed scrap metal since the 1950s was the modern-day Frito Lay (DAR No. 31) site in English Kills (see Table 5-3, Figure 5-1, and Appendix B [specifically, the relevant upland site summary] of the draft DAR, as well as the updated upland site summary, in Appendix J of this RI Report). The Lipsett Steel scrap metal processing facility existed on the Frito Lay site from 1951 to 1983 (Claster 2000). Lipsett Steel shipped and received scrap metal by barge. The company had four diesel cranes with buckets and electronic magnets and eight diesel crawler cranes used in the rear yard. The site had an open storage area for up to 24,000 tons of scrap metal (USACE 1965). Two other scrap metal processing facilities operated at the Frito Lay site from 1983 to 2006. During an RI in 2010, surface soil sampling was conducted within and along the banks of the Frito Lay site for PCBs, metals, VOCs, SVOCs, and pesticides. PCBs were detected in site soils with concentrations as high as 3,200 milligrams per kilogram (mg/kg). Concentrations of metals, including, but not limited to, Pb, Hg, and arsenic (As), were also detected. PAHs were detected in soil and groundwater samples. During 2012 and 2013 wet weather surveys, shoreline banks at the Frito Lay site were found to be erodible. Remedial activities occurred in 2013. More than 17,000 tons of Toxic Substances Control Act (TSCA) and non-TSCA PCB-impacted material was excavated at depths of up to 12.5 feet below ground surface and disposed of off site. Soils with PCBs up to 10 mg/kg were capped in place. These findings indicate this site was at least a historical source of TSCA-level PCBs, metals, and PAHs to English Kills (see the relevant upland site summary in Appendix B of the draft DAR in Appendix J of this RI Report).

The Hugo Neu (DAR No. 125) site (see Table 5-3, Figure 5-1, and Section 7.5 of the draft DAR in Appendix J), located on the Queens side of Newtown Creek (just upstream of the mouth of Dutch Kills), has also operated as a scrap metal and recyclables yard since approximately 1970 (Newtown Creek Properties 1970). Along the site's bulkhead, there is room for up to four hopper barges. Scrap metal is trucked to the facility, sorted, and shipped by barge to the larger Hugo Neu facility in Jersey City for additional processing or reuse (see the relevant upland site summary in Appendix B of the draft DAR, as well as the updated upland site summary, in Appendix J of this RI Report; NYSDOT and FHWA 2005). Hugo Neu purchases scrap metal from other independent scrap yards, including Charles J. King, Inc. (DAR No. 206), which is located on the east bank of English Kills. Hugo Neu also provides maritime transportation of scrap metal for Charles J. King. Similar to other metal scrappers described in this section, Hugo Neu is a source of PCBs to Newtown Creek, as elevated concentrations of PCBs were measured in stormwater samples collected from the site during Phase 2 (see Section 5.1.3.3.4). A large plume of NAPL is also present on the north side of the site near Dutch Kills (see the relevant upland site summary in Appendix B of the draft DAR, as well as the updated site summary, in Appendix J of this RI Report). Charles J. King also had one

NYSDEC-documented spill (NYSDEC Spill No. 9309269) in 1993, of an unknown amount of PCB oil to Newtown Creek. The spill record was closed the next day (NYSDEC 2016a) (see Table 3-1, as well as the relevant updated site summary in Appendix J).

3.2.6.13 Paints and Pigments Industry

The paints and pigments industry includes the mixing and sales of paint, varnish, dry colors, pigments, acids, and heavy metals (e.g., Cd, Cr, Cu, Pb, and zinc [Zn]) (America's Corporate Foundation 1928). Paint factories began to appear in industrial centers in the mid-1880s. Due to the weight of the prepared paint and challenges with its transportation, a decentralized structure of small manufacturers dominated the industry until the mid-1900s (American Coatings Association 2019). Of the metals, Pb has been used most extensively in the paint pigments (e.g., Pb tetroxide – red Pb, Pb (II) chromate – chrome yellow, and Pb (II) carbonate – white Pb), with some old paint containing as much as 50% Pb (ATSDR 2007). With the finding of adverse health effects from Pb exposure, paint manufacturers started replacing Pb pigments in some paints with safer alternatives before World War II. Common house paints have contained little, if any, Pb since the 1950s, and in 1978, the Consumer Product Safety Commission banned the use of Pb in consumer paint. The operation of paint and pigment factories began to slow by 1978, with the enforcement of compliance of major air pollution sources (USEPA 1978), and the majority of large-scale paint and pigment factories had virtually disappeared from Newtown Creek by the early 2000s.

3.2.6.13.1 Contaminants Associated with Paints and Pigments Industry

According to the 1990 USEPA *Guides to Pollution Prevention* (USEPA 1990a), paint manufacturing facilities generated large quantities of hazardous and non-hazardous wastes, including cyanides, waste solvents, equipment cleaning wastewater, paint spills, leftover containers, and pigment dusts from air pollution control equipment (Haynes 1932). These waste streams would have included VOCs, SVOCs (including PAHs), CVOCs, PCBs, heavy metals, and other inorganics from wastewater treatment effluents. Petroleum distillates are essential ingredients for paint, varnish, and dye manufacturing (Hornix 1992; Regional Survey 1924). Pigments (often containing metals) and solvents were also important raw materials for lacquer and paint; and up until the 1979 ban, PCBs were often used as plasticizers in paints. Because PCBs are persistent in the environment, the manufacturing of paints would not be the sole PCB source from this industry; paints applied to buildings and materials containing PCBs could also reach the creek by becoming entrained in stormwater runoff (either as paint chips or by leaching from the paint into the stormwater).

3.2.6.13.2 Paints and Pigments Industry Sites near Newtown Creek

E.I. Du Pont de Nemours and Company was the first manufacturer to produce paints in the vicinity of Newtown Creek in 1912. In 1918, a prominent chemical company, Whittaker, Clark, and Daniels, Inc., moved their manufacturing plant from NYC to the Duveen Soap Corp. site (no DAR No.) near the head of English Kills (*Paint Industry Magazine* 1952; USACE 1932). The history of paints and pigments in the region followed a similar pattern for the next 35 years, and by 1963, more than two dozen paint and pigment manufacturers were present throughout Brooklyn (NYSID 1963). Although, as noted in Section 3.2.6.13, most of the paint and pigment manufacturing facilities had disappeared from Newtown Creek by the early 2000s, including Empire State Varnish Company, Inc., which was demolished in 2008 and is now the site of an office building.

3.2.6.14 Paper Products Industry

Pulp and paper production was a thriving industry in the area in the late 1800s and early 1900s (USEPA 1976b). Between 1902 and 1920, approximately 12 paper producers erected paper-manufacturing plants between Brooklyn and Queens. The highest concentration of paper manufacturers was in the Long Island City community north of Newtown Creek. The paper industry in this vicinity included the manufacture of paper bags and boxes, stencil papers, boxboard and lining, writing papers, and tablets (CCBQ 1920). Paper products were derived from three main materials: approximately 80% cellulosic raw materials (wood pulp), 20% waste paper, and less than 1% synthetic materials, such as rags or inorganic asbestos fibers (USEPA 1976b).

3.2.6.14.1 Contaminants Associated with Paper Products Industry

The process of manufacturing paper products was, and continues to be, a multistage process requiring raw material harvesting, wood preparation, pulping, bleaching, stock preparation, sheet formation, pressing, and drying. The pulping and bleaching processes require huge amounts of freshwater and produce large amounts of wastewater with non-biodegradable contaminants (USEPA 1976b). Wastewater contaminated by both organic and inorganic chemicals, including lignin, phenols, sulfides, and PAHs, as well as PCBs from carbonless copy paper, was released by paper manufacturers into waterbodies through municipal sewer systems (USEPA 1976b). With the exception of PCBs via the recycling of carbonless copy paper, which was not introduced until after the paper mills in Newtown Creek shut down, these contaminants may have been discharged directly from the facilities and/or through municipal sewers into Newtown Creek.

3.2.6.14.2 Paper Products Industry Sites near Newtown Creek

Two paper manufacturers in the Newtown Creek area were the Centaur Paper Mills Supply Corporation, which operated from 1938 to circa 1951 at the Masluf Realty Corp. (DAR No. 201) site, and the Alfred Bleyer & Co. Paper Products, which operated along East Branch at the Western Beef Properties (DAR No. 212) site from 1920 to 1950 (NYT 1938; USACE 1942; Sanborn 1951; Serant 1995, and the updated upland site summary for DAR No. 212 in Appendix J).

Though it was clear that paper facility wastes required special treatment at water treatment plants, it was not until 1949 when the New York governor executed the New England Interstate Water Pollution Control Compact on behalf of New York State (NYCDPW 1957) that pretreatment facilities were installed.

3.2.6.15 Petroleum Refining and Bulk Storage[35]

This section discusses generally the petroleum refining process and the associated COPCs that can be generated during the refining and storage process. Additionally, this section addresses the development of petroleum operations along Newtown Creek specifically, including a broad overview of some of the historical and current petroleum operations near the creek.

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[35] ExxonMobil; Texaco, Inc.; and BP Products North America, Inc., prepared the historical industry summaries for their sites in Section 3.2.6.15 and subsections, and the industry information and site status summary are according to these companies.

3.2.6.15.1 General Overview of Petroleum Refining Industrial Process

Petroleum refining is the physical, thermal, and chemical separation of crude oil into its major distillation fractions, which are then processed through a series of separation and conversion steps into finished petroleum products. The primary products of the industry have included fuels (e.g., motor gasoline, diesel, jet fuel, kerosene), finished nonfuel products (e.g., solvents, lubricating oils, greases, asphalt), and chemical industry feedstocks (e.g., naphtha, ethane, propane, ethylene, benzene, toluene, xylene). The primary feedstock used in petroleum refining is crude oil, which is a complex mixture of hydrocarbons and impurities. The crude oil refining process is highly variable and depends largely on the composition of the crude oil and the desired end-products. A key step in the refining process is distillation of crude oil into its various fractions using processes such as heating, vaporization, fractionation, condensation, and cooling. Following distillation, additional downstream processes are employed, such as cracking, hydrotreating, polymerization, solvent extraction, dewaxing, and other operations. These processes change the molecular structure of the hydrocarbons by breaking them down into smaller constituents, joining them to form larger molecules, or transforming them into different compounds. Other activities can include product blending and the addition of chemical additives (USEPA 1995d).

In the early years of petroleum production, the basic distillation process involved boiling the crude oil in stills, as a batch process. For each batch, the key distillation products were drawn off sequentially in order of their boiling point range (e.g., naphtha, straight-run gasoline, kerosene, fuel oils, and lubricating oils), then condensed, captured, and conveyed to storage facilities (Haney 1923). In the late 1800s, crude oil was primarily refined into kerosene, which was widely used in lamps and stoves. Following distillation, refineries typically treated the kerosene to improve the color and odor and to remove the more volatile constituents. By the early years of the twentieth century, development of the internal combustion engine shifted demand of petroleum products to automobile fuels and lubricants.

3.2.6.15.2 Contaminants Associated with Petroleum Refining and Bulk Storage

Petroleum refining and storage operations produce a wide range of waste types and contaminants depending on the crude oil source, the refining processes used, and products produced. Toxic release inventory records for the refining industry report that more than

100 different types of chemicals are released or transferred via spills and/or air emissions during refining operations (USEPA 1995d). Some of the most commonly reported chemicals include aromatic hydrocarbons (e.g., BTEX). A range of other chemicals may also be associated with petroleum operations, including additives that are historically or currently blended with the petroleum, such as methanol, ethanol, alkyl Pb compounds, and methyl tertiary butyl ether, as well as chemical feedstocks such as propylene, ethylene, and naphthalene. Inorganic compounds including various metals, ammonia, and spent sulfuric acid are also common components of refining wastes. Crude oil contains a class of multiple-ring, high-molecular-weight compounds called PAHs. PAH compounds do not constitute a large fraction of crude oil, but are relevant contaminants associated with petroleum operations due to their toxicity (Kostecki and Behbehani 1995). Under some circumstances, NAPL may be associated with releases from operations associated with the petroleum refining and bulk storage industry.

3.2.6.15.3 The Development of Petroleum Operations at Newtown Creek

In the mid-1800s, Newtown Creek served as a natural hub for commercial and industrial development due to its access to a navigable waterway. Between 1880 and 1900, improved adjacent street networks and added rail and lighterage services supported further industrialization of the creek. Because the waterway could support large vessels, new industrial operations evolved to include numerous petroleum refining operations.

The discovery of crude oil and drilling of the first petroleum production well in Titusville, Pennsylvania, in 1859 led to an increased demand for petroleum lamp oil (Petroleum Publishing, Inc. 1959). After that time, in addition to being used for kerosene production, petroleum distillates were essential to paint, varnish, and dye manufacturing (Hornix 1992; Regional Survey 1924). Moreover, multiple ancillary industries were also connected to the development of the petroleum industry. Chemical manufacturers, for example, produced chemicals used in the refining process, whereas fertilizer manufacturers used residual sludge from the refining process (BPL 1886; Baker and Kent 1887).

By 1892, three operating oil refineries were located on Newtown Creek: 1) Olephane Oil Works (shut down in 1905); 2) Kings County Oil Works (also known as Sone & Fleming

Works and the Brooklyn Refinery); and 3) Central Refining Oil Works (sold and consolidated into the Brooklyn Refinery in 1900). In addition, there was the POW (now Waste Management of NY/Steel Equities [formerly POW] [DAR No. 56]), which primarily operated as a paraffin plant (closed in 1949).

The Brooklyn Refinery, the last petroleum refinery operating on Newtown Creek, ceased operations in 1965 and converted to a petroleum products bulk storage and transfer terminal. It, along with other petroleum product terminals, engaged in receiving and dispatching bulk shipments of a range of refined petroleum products from pipelines, tankers, railcars, barges, and trucks for subsequent distribution. Products received and distributed by petroleum terminals likely changed over time as refining industry products and consumer demands evolved.

Potential sources of petroleum releases in the early years may have included releases from ASTs, USTs, and underground and aboveground pipes used to move product around the operations. Additionally, potential petroleum releases may have resulted from filling and withdrawal, and blending, as well as loading and unloading of product during shipping and barging operations, cleaning of tanker trucks and railcars, runoff from operational and secondary containment areas, leaks, and accidental spills (see Section 3.2.11).

3.2.6.15.4 Summary of Ownership History and Petroleum Operations at Newtown Creek

Petroleum refining and storage operations along Newtown Creek were conducted by various entities throughout the creek’s long industrial history.

This section includes a discussion of the historical ownership and a summary of operations for the following significant petroleum/oil facilities identified in the draft DAR. A detailed ownership history is provided in the draft DAR (see Sections 5.6 and 7 and Appendices B and C of the draft DAR in Appendix J of this RI Report).

ExxonMobil Greenpoint Remediation Project (DAR No. 53) (see Section 5.6.6 of the draft DAR in Appendix J)

In 1892, Standard Oil Company of New York (SOCONY) acquired from Sone & Fleming Manufacturing Company, Limited, parcels of land and refining equipment that later became known as the Brooklyn Refinery. Remaining parcels comprising the Brooklyn Refinery, owned by others as of 1892, were acquired by SOCONY at later points in time. The Brooklyn Refinery, which spanned the area from Norman Avenue and North Henry Street to Newtown Creek, was operated by SOCONY, which later became Mobil Oil Corporation (Mobil). A 1942 USACE survey shows that the Brooklyn Refinery had an operating capacity of 19,000 barrels (798,000 gallons) per day and a storage capacity of 875,000 barrels (36,750,000 gallons), and that one 12-inch, three 6-inch, and four 8-inch pipelines extended from the dock to the storage tanks on site (USACE 1942). A 1953 USACE survey notes that the refinery included approximately 200 steel storage tanks, as well as several pipelines extending from the dock to the storage tanks (USACE 1953). Refining operations at Mobil's Brooklyn Refinery continued until 1965. After refinery operations ceased, portions of the site continued to be used by Mobil for petroleum bulk storage. Mobil used the property it retained as a petroleum bulk fuel storage and distribution terminal until 1993. The terminal structures were demolished by 2007.

BP Products N America Brooklyn Terminal (DAR No. 48) (see Section 5.6.3 of the draft DAR in Appendix J)

After operations ceased at the Mobil Brooklyn Refinery in 1965, significant portions of the property were sold to various companies for continued industrial use. In 1968, American Oil Company (Amoco) purchased an approximately 10-acre parcel located at the northwest corner of Apollo Street and Norman Avenue. This property began operating as an oil storage and distribution terminal in 1970. Following the 1998 Amoco-BP merger, the facility was operated by BP. In 2016, the terminal was sold to Kinder Morgan, who continues to operate the facility as an oil storage and distribution terminal, storing finished petroleum products and additives including diesel fuel, No. 2 fuel oil, kerosene, gasoline, and ethanol. The property consists of aboveground product storage and chemical additive tanks, one diesel UST (used for truck refueling), two loading racks, and multiple single-story concrete block buildings. The storage capacity of the terminal is 5,902,512 gallons.

Waste Management of NY/Steel Equities (formerly POW) (DAR No. 56) (see Section 5.6.7 of the draft DAR in Appendix J)

On the Queens side of Newtown Creek in 1879 and 1887, Charles Pratt and Company acquired property that became known as the POW. The POW operated primarily as a paraffin plant manufacturing wax and, at times, lubricating oils. SOCONY acquired Pratt's facility in 1892. SOCONY ceased operations at the POW in approximately 1949, and the property was subsequently sold and redeveloped for various industrial activities including, but not limited to, petroleum, chemical, and gravel manufacturing and warehousing/storage operations. Information on historical spills from the site is included in Section 3.2.11.

Empire Merchants/Former Paragon Oil Terminal (DAR No. 200) (see Section 5.6.2 of the draft DAR in Appendix J)

The Paragon Oil Company, and subsequently Texaco, operated a petroleum products bulk storage and distribution terminal from 1934 to 1968, on an 11-acre site bordered by Meeker Avenue to the east and Bridgewater Street to the south. The former terminal stored and distributed fuel oils (Nos. 2, 4, and 6), lubricating oil, leaded gasoline, diesel fuel, and kerosene (see Section 5.6.2, Table 5-3, and Appendix C [specifically, the relevant Respondent site summary] of the draft DAR in Appendix J of this RI Report). According to historical Sanborn maps, the former bulk terminal operation included a varying number of ASTs and USTs of differing capacity through time. Terminal operations peaked in the 1960s with a storage capacity of 6.1 MG of finished petroleum product. More than 64% of tankage was used for the storage of Nos. 4 and 6 fuel oil, and lubricating oils. After terminal operations ceased in 1968, the property was sold to Peerless Importers. The property is currently operated by Empire Merchants as a series of interconnected warehouses that cover approximately 8 acres of the property, with the remainder of the property comprising truck and car parking.

Getty Terminals Corp. #58220 (DAR No. 47) (see Section 7.3 of the draft DAR in Appendix J)

Getty Oil started operating on Newtown Creek in the 1930s as the Tidewater Oil Company (Tidewater) at 30-23 and 30-21 Greenpoint Avenue, Queens. Tidewater utilized the site as a gasoline storage and distribution center. A 1953 USACE survey indicated that two 8-inch and two 4-inch pipelines extended from the dock to 11 steel storage tanks, with a total capacity of 20,000 barrels (840,000 gallons) (USACE 1953). Tidewater and its parent corporation, Getty Oil, operated the site until 1984, when the property was sold to Power Test Corp (unrelated to Getty Oil). LukOil acquired the site in 2001 and operated petroleum storage tanks, pipelines, truck loading racks, and ancillary equipment.

Malu Properties/Former Ditmas Oil/Former Gulf Oil (DAR No. 123) (see Section 5.6.8 of the draft DAR in Appendix J)

Gulf Oil operated the Gulf Refining Co. site located at 364 Maspeth Avenue along Newtown Creek as a bulk terminal facility from 1945 to 1985. The site was sold in 1985 to Termynx/Ditmas Oil, who continued to operate the 7-acre site as a petroleum products storage and distribution bulk terminal. The site received various petroleum products from barges in both the adjacent waterway and the Buckeye Pipeline that transected the site (see Section 5.6.8 and Appendix C [specifically, the relevant Respondent site summary] of the draft DAR in Appendix J of this RI Report). Petroleum was stored in ASTs and USTs and then distributed into trucks through an underground piping system. A 1965 USACE survey identified four pipelines operated by Gulf Oil that connected from the wharf to 21 ASTs and USTs with a total capacity of 1,836,000 gallons. In 1985, the total storage capacity was 3,745,000 gallons. While gasoline was the main product stored at the facility, kerosene, diesel, and No. 2 fuel oil could also be found in smaller tanks. Petroleum storage operations at the site were decommissioned between February 2004 and June 2005. The site is currently owned by Malu Properties, Inc., and is operated by the NYC Marshals as an automobile impound lot. See Section 3.2.11 for information on spills.

Motiva Brooklyn Terminal (DAR No. 50) (see Section 7.1 of the draft DAR in Appendix J)

In 1930, Shell Eastern Petroleum Products Inc. (also known as Shell Oil Company [Shell]), purchased its 7.5-acre site along Newtown Creek from Forest Box and Lumber Company. The site, located at 25 Paidge Avenue, Brooklyn, functioned as a petroleum receiving station with both pipelines and USTs. The site featured four 6-inch pipelines, and 40 USTs and 10 ASTs, totaling approximately 40,950 barrels (1,719,000 gallons) (USACE 1953). According to a 1959 Water Usage Survey, Shell stated that the terminal handled approximately 100 MG of petroleum products annually (Hazen and Sawyer 1959). A 1965 survey showed two additional storage tanks that had been added to the facility, bringing its total capacity to 63,810 barrels (2,680,020 gallons) (USACE 1965). In August 1998, Motiva Enterprises, LLC (Motiva), notified NYSDEC that Motiva would own and operate some Shell assets, including

Shell's Brooklyn Terminal. Since purchasing the site from Shell in 1998, Motiva has continued on-site operations. The site receives, stores, and transfers petroleum products (such as diesel; Nos. 1, 2, and 4 fuel oil; and gasoline). Products are stored in ASTs and USTs and are moved to and from the facility via pipeline, barge, and tanker truck.

Metro Terminal (DAR No. 52) (see Section 7.2 of the draft DAR in Appendix J)

Metro Terminal, which occupies approximately 4.5 acres adjacent to Newtown Creek at 498 Kingsland Avenue, Brooklyn, is a multi-million-gallon petroleum storage facility and transfer terminal. It has been in use as an oil terminal since at least 1916 and has been the main office for Metro Energy since 1986. A 1953 USACE survey indicated that the company used its dock to receive petroleum products by barge, then transported by one 10-inch, one 6-inch, four 5-inch, and three 4-inch pipelines from the dock to 18 steel storage tanks with a total capacity of 50,000 barrels (2,100,000 gallons) (USACE 1953).

A 1965 USACE survey indicated that Ross Oil operated the dock, which included eight 6-inch pipelines extending to ASTs and USTs with a total capacity of 25,070 barrels (1,052,940 gallons) to receive petroleum products (USACE 1965).

A 1978 USACE survey indicated that the company operated two docks with three 6-inch pipelines that connected with other pipelines extending to five steel storage tanks with a total capacity of 88,300 barrels (3,708,600 gallons) (USACE 1978). Metro Terminal continues to operate and receive petroleum products and additives, including diesel, biodiesel, xylene, gasoline, and fuel oil. The facilities include a truck loading rack, fueling area, maintenance building, 11 ASTs, and 13 USTs.

B.C.F. Oil Refining, Inc. (DAR No. 27) (see Section 7.8 of the draft DAR in Appendix J)

The B.C.F. Oil site is a 1.9-acre site located adjacent to English Kills. From at least 1932 to 1979, the site was an oil terminal with various occupants, including Atlantic Basin Oil Co., Morania Oil Co. Inc., and AR Fuels (USACE 1932). As of 1959, Morania Oil handled No. 2 fuel oil, kerosene, and gasoline (Hazen and Sawyer 1959). By 1965, the company had a total storage capacity across the 12 USTs and four ASTs of 27,380 barrels (1,149,960 gallons) (USACE 1965). In 1979, the site was modified for waste oil processing. From 1980 to 1994, B.C.F. Oil and Calleia Brothers Inc., used the site for waste oil recycling operations. As of 1982, the processing capacity was approximately 48,000 to 50,000 gallons per day (Calleia 1982). The site had 11 USTs and four ASTs with a total storage capacity of approximately 792,000 gallons. Currently, the NYC Marshals use the site as an automobile impound lot.

Morgan Oil Terminal, Brooklyn (DAR No. 60) (see Section 7.10 of the draft DAR in Appendix J)

Morgan Oil Terminal operated from 1982 to 1994 as an oil terminal and distribution facility on a 3.52-acre site adjacent to English Kills. Oil operations started at the site in the 1940s with prior site owners, including Coal Heat & Fuel Oil, Inc., and Premium Coal and Oil Co., Inc. (see Section 7.10 of the draft DAR in Appendix J). Historically, the site operated as a brick and lime storage facility, asphalt manufacturer, coal-storage and distribution facility, lumber yard, metal incinerator, and wire/sheet metal manufacturer. Coal-storage contamination is discussed in detail in Section 3.2.6.5.

A 1965 survey indicated that Premium Coal and Oil Co., Inc., had seven steel storage tanks with a total storage capacity of 46,024 barrels (1,933,008 gallons) (USACE 1965). RIs at the site began in 1993 and continue to the present day. As of 2010, the site was undergoing active remediation for potential future redevelopment. See Section 3.2.11 for information on spills.

Bayside Fuel Oil Depot – 1100 Grand St (DAR No. 51) (see Section 7.11 of the draft DAR in Appendix J)

In 1996, Bayside Fuel Oil Depot was established on a 1.5-acre site adjacent to English Kills to be used for the storage and distribution of Nos. 1, 2, and 4 fuel oil and diesel fuel. Prior to this and since the 1910s, the site had been utilized for the storage and distribution of coal and petroleum products including diesel; kerosene; and Nos. 1, 2, 4, and 6 fuel oils. Coal-storage contamination is discussed in detail in Section 3.2.6.5. Portions of the site may also have been used for concrete block manufacturing and food and ice businesses. Previous tenants of the site include Cirillo Brothers Fuel Oil; Filtered Fuel Oil Corp.; Mishkin Brothers Coal Co.; United Cement Products Co., Inc.; and American Ice Co. Bayside Fuel Oil Depot continues to operate and has a total storage capacity of 1,651,241 gallons in five USTs and one AST (see Section 7.11 of the draft DAR in Appendix J).

Buckeye Pipeline Facility (DAR No. 106) (see Section 7.4 of the draft DAR in Appendix J)
The Buckeye Pipeline Facility is located on a 1-acre site adjacent to Newtown Creek. Beginning operations in 1975, Buckeye Pipeline Facility currently uses the facility to transfer petroleum products via underground pipes to or from oil terminals in Greenpoint. The products being transferred include fuel oil, gasoline, and naphtha. Prior occupants of the Buckeye Pipeline Facility site since 1900 include the Long Island Railroad Co., Tidewater, Long Island Pipe Line Corp, and Rosil Realty Corp. See Section 3.2.11 for information on spills.

Sunoco, Inc., and Predecessors (No DAR No.)
Sunoco, Inc., operated an oil terminal and distribution facility on a 1.4-acre site adjacent to Newtown Creek from 1923 to 1980. Located at 53-02 11th Street, Long Island City, the site was utilized as a petroleum bulk terminal for the receipt, storage, blending, and distribution of petroleum products. In addition, vehicle maintenance of the distribution trucks also occurred on site. A 1942 USACE survey reported that oil and gasoline was delivered to 38 tanks (total capacity of 690,000 gallons) by one 6-inch pipeline and four 4-inch pipelines (USACE 1942). A 1979 survey indicated that the site handled and stored 682,000 gallons of gasoline; 50,400 gallons of kerosene; and 136,200 gallons of lube oils. In the same survey, it was also indicated that the site did not use an oil-water separator, and instead stormwater was managed via the municipal sewers (Binswanger/Herman Company 1979).

3.2.6.16 Plastics Industry

The plastics molding and forming industry (PM&F) is a large and diversified industry that uses plastic materials to produce a wide variety of consumer and industrial products. The product markets include automobiles, appliances and electronics, construction materials, disposables, household furnishings, housewares, and medical products. There are many different types of production processes that process various combinations of raw materials. PM&F plants blend, mold, form, or otherwise process plastic materials into intermediate or final plastic products. They include commonly recognized processes such as extrusion, molding, coating and laminating, thermoforming, calendering, casting, foaming, cleaning, and finishing. In addition, the plastics industry also includes the manufacture of products or product groups such as synthetic fibers (e.g., rayon fibers), thermoplastic resins, thermosetting resins, commodity organic chemicals, bulk organic chemicals, and specialty organic chemicals.

Process water is used in the plastics industry to cool or heat plastic products, to clean the surfaces of both the plastic products and the equipment used to produce those products, and to finish plastic products.

3.2.6.16.1 Contaminants Associated with Plastics Industry

Pollutant outputs generated by plastics industries generally include VOCs, contaminated polymers, plasticizers, chemical additives used to increase the plasticity and fluidity of materials, and wastewater from equipment cleaning. Manhattan Poly Bag (DAR No. 130) included three tar tanks (Sanborn 1933) (tar is used for carbon fiber manufacturing), and elevated PCBs have been found in sediments adjacent to the site (see the relevant upland site summary in Appendix B of the draft DAR, as well as the updated upland site summary, in Appendix J of this RI Report). Specific pollutants and contamination pathways vary, depending on which types of plastics are being processed and which type of process is being used (see Table 5-3 of the draft DAR in Appendix J for Former NuHart Plastic Manufacturing [DAR No. 29], Kalex Chemical Products, Inc. [DAR No. 127], and Manhattan Poly Bag [DAR No. 130]; USEPA 1997a). Given the large number of different processes used in the plastics industry, it is difficult to estimate the types and quantities of contaminants that may have been discharged from plastics industry sites near Newtown Creek.

3.2.6.16.2 Plastics Industry Sites near Newtown Creek

The plastics industry has historically occupied a number of sites in the vicinity of Newtown Creek, particularly in Brooklyn (see Table 5-3 and Figure 5-1 of the draft DAR in Appendix J). Former NuHart Plastic Manufacturing (DAR No. 29) was a plastic and resin manufacturer that operated from 1940 to 2004 (Short 2010); Kalex Chemical Products, Inc. (DAR No. 127), was an unlaminated plastics and vinyl films manufacturer that operated from 1978 to 1997; and Manhattan Poly Bag (DAR No. 130) was a plastics, foil, and coated paper bags manufacturer that operated from 1986 to 2007 (see the relevant upland site summaries in Appendix B of the draft DAR, as well as the updated upland site summary for Manhattan Poly Bag, in Appendix J of this RI Report).

3.2.6.17 Printing

By 1860, the printing industry was firmly established in Greenpoint (Felter 1919). By 1916, several printing and publishing companies were operating within (or near) the vicinity of the Study Area, including imprinting, bookbinding, stationery, print design, commercial printing, graphics, copies, custom printing, and color printing (CCBQ 1920). In general, the printing industry involves five different processes: lithography, gravure, flexography, letterpress, and screen printing. Different raw materials, including inks, solvents, and adhesives, are employed depending on the process.

3.2.6.17.1 Contaminants Associated with Printing Industry

The three major types of wastes in the printing industry include solid wastes, wastewater, and air emissions. Solid wastes may consist of empty containers, used film packages, outdated materials, damaged plates, developed film, bad printing or spoilage, damaged products, and scrap paper. Wastewater may contain lubricating oils, waste ink, cleanup solvents, photographic chemicals, acids, alkalis, and plate coatings, as well as PCBs and metals such as silver (Ag), iron (Fe), Cr, Cu, and barium (Ba). Printing operations also produce VOC emissions from the use of cleaning solvents and inks, as well as alcohols and other wetting agents (ISTC 2016). Given the large number of processes that may be used in the printing industry, it is difficult to estimate the types and quantities of contaminants that may have been discharged from printing industry sites near Newtown Creek.

3.2.6.17.2 Printing Industry Sites near Newtown Creek

A number of sites associated with printing are in the vicinity of Newtown Creek and remain in operation to the present day. Architectural Coatings, Inc. (DAR No. 103), is a current printing and binding company located in Brooklyn. The site was also historically occupied by the Print House, which specialized in printing and engraving. Confort & Company Inc. (DAR No. 6) is also a current printing company located in Long Island City (see Table 5-3, Figure 5-1, and Appendix B [specifically, the relevant upland site summaries] of the draft DAR in Appendix J of this RI Report).

3.2.6.18 Railyards

In the early 1870s, LIRR filled and bulkheaded approximately 2,000 feet along the creek west of Dutch Kills and constructed the Wheelspur Yards adjacent to Newtown Creek (NYSDOT and FHWA 2005). By 1897, LIRR tracks had been constructed in Queens along the length of the creek (the Montauk Division) and east of the creek in Brooklyn Bushwick Line (near present-day English Kills and East Branch, between Meserole Street and Randolph Street; USGS 1898). By 1957, LIRR operated at least seven yards near Newtown Creek (Terminal Yards, Arch Street Yards, Wheelspur Yards, Blissville/Greenpoint Avenue Yards, Maspeth Yards, Varick Avenue Freight Yard, and Bushwick Yards) (USGS 1947, 1956).

In 1903, the Pennsylvania Tunnel and Terminal Company, a subsidiary of the Pennsylvania Railroad Company, acquired land in Long Island City for the Sunnyside Yard. The Sunnyside Yard was built from 1907 to 1910, with operations officially started on November 27, 1910 (ICC 1918). Pennsylvania Railroad Company owned the property for the first six decades of the site's history before merging with New York Central in 1966 to form the Pennsylvania New York Central Transportation Company (Moody's 1967). On April 1, 1976, the Consolidated Rail Corporation acquired the site and the same day conveyed it to the National Railroad Passenger Corporation (Amtrak; NYSDEC 2010a; Amtrak 2012).

3.2.6.18.1 Contaminants Associated with Railyards

Historical operations at railyards involved the use of coal-fired locomotives, boilers, and incinerators resulting in the production of large amounts of cinders and ash, which were often used as on-site fill. Cinders and ash would have been sources of PAHs and metals. Coal-storage contamination is discussed in detail in Section 3.2.6.5.

As railyards transitioned from coal-fired power to electricity and diesel-powered locomotives, mobile transformers were placed on trains and stationary transformers were installed in the yards (Amtrak 2012). Specifically, in 1949, there were 49 stationary transformers at the Amtrak Sunnyside Yard, 14 of which contained fluid with greater than 50 parts per million PCBs (Roux 2001). Railyards also stored bulk petroleum products (e.g., diesel, fuel oils) for fueling and maintaining locomotives, and coach cleaning operations also occurred at many railyards, including the Sunnyside Yard (JESP 1911).

Often CVOCs are observed in rail maintenance areas, because they were used as degreasers (USEPA 2002a, 2006b). In addition, PAHs may leach from creosote-treated railway ties. Slag was often used as railbed ballast and could have contained PAHs and metals. Spills from railroad operations, and the loading and unloading of railcars, could have also been sources of contaminants to the creek.

3.2.6.18.2 Railyards near Newtown Creek

Two historical LIRR railyards (Long Island City Freight Yard [AOC 1; DAR No. 128] and Long Island City Freight Yard [AOC 2; DAR No. 129]) and the Amtrak Sunnyside Yard (DAR No. 102) are listed in the draft DAR (see Table 5-3 and Figure 5-1 of the draft DAR in Appendix J). Releases have occurred at the Amtrak Sunnyside Yard (DAR No. 102), associated with historical and recent fueling operations and maintenance activities, most commonly related to broken machinery or tank failures (see the relevant upland site summary in Appendix B of the draft DAR in Appendix J of this RI Report). At least 33 NYSDEC spills have been documented at the site, 7 of which were documented between 1986 and 1999 as PCB oil or transformer oil containing PCBs. Many more PCB-related spills likely occurred earlier in the site history that were not documented, because a formal spill prevention and reporting system had not been established by New York State before that time. In the 1980s, NYSDEC listed the Amtrak Sunnyside Yard as a “Class 2” site in the State Registry of Inactive Hazardous Waste Disposal Sites. As a result of this listing, NYSDEC issued an Order on Consent in 1989 between NYSDEC, Amtrak, and New Jersey Transit Corporation No. W2-0081-87-06, which required Amtrak to conduct RIs. TSCA-level PCBs (Aroclors 1254 and 1260) were detected in soil, as well as in sediment from the site sewer system (Aroclors 1248, 1254, and 1260). Up to 4.4 micrograms per liter (μg/L) total PCB (Aroclors 1260, 1254, and 1248) were detected in the primary (combined) sewer system wastewater discharge. Four distinct groundwater plumes were identified at the site, including a plume of NAPL containing PCBs (Aroclors 1248 and 1260) located at the northern portion of the site.

Currently, the sewer system at the Amtrak Sunnyside Yard consists of two separate subsystems. The "primary" system is a combined sewer designed to receive sewage and stormwater drainage from the majority of the site and ultimately discharges to the Bowery Bay WWTP via the upper level interceptor. The "secondary" system, which serves a relatively small area of the western portion of the site, was designed to receive only stormwater. This system drains to Dutch Kills via BB-610 (Roux 2012; NYCDEP 2013b) and was sampled during the point sources sampling program. The point sources sampling program and results are discussed in Section 5.1.3 of this RI Report and Section 2 of Appendix E.

3.2.6.19 Sawmills and Lumberyards

With the appearance of shipyards (see Section 3.2.6.20) and the demand for houses came lumberyards and dealers in building materials. The first lumberyard was established in 1850 at the foot of Kent Street (Felter 1919). After the Civil War, Leary's Lumber Company and shipyard, located on Whale Creek, produced not only pilings, but also dredges, scows, floating derricks, and car floats (Harding 1944). In the 1880s, several lumber companies occupied the north shore of the creek (Goodwin and Associates 2012).

Active lumberyards declined after 1950 (Goodwin and Associates 2012; NYSDOT and FHWA 2005), but the storage of lumber on upland sites is still occurring today (e.g., on East Branch at Felman Metropolitan [DAR No. 217] and Mione Transit Mix).

3.2.6.19.1 Contaminants Associated with Sawmills and Lumberyards

Typical contaminants associated with sawmills and lumberyards include VOC emissions and chemicals included in older wood preservatives, such as metals (Cu, Cr, and As), pentachlorophenol, and PAHs associated with creosote-treated wood (USEPA 2016a). Given the varied nature of historical sawmill and lumberyard sites near Newtown Creek, the types and quantities of contaminants that may have been discharged from these sites into Newtown Creek are not conclusively known.

3.2.6.19.2 Sawmills and Lumberyards near Newtown Creek

Between 1860 and 1950, lumberyards were a prominent industry on the creek (Goodwin and Associates 2012). Eleven lumberyards were documented by the War Department in the

1930s (War Department 1930b), including the J.P. Williams & Company lumberyard located at the Getty Terminals (DAR No. 47) site (1898 to 1936); the Motiva Brooklyn Terminal (DAR No. 50), occupied by the Forest Box and Lumber Co.; the Hardy Voorhees & Co. Lumber Yard and Planing Mill at the Amboy Bus Co. (DAR No. 211) and Western Beef Properties (DAR No. 212) sites; and the Morgan Avenue Kindling Wood Works sawmill at the Morgan Oil (DAR No. 60) site (see Table 5-3 and Figure 5-1 of the draft DAR in Appendix J; Sanborn 1908, 1914).

3.2.6.20 Shipbuilding

Shipbuilding was a large industry on the shores of Newtown Creek that peaked in the mid-1800s. The vast majority of the ships built were of wooden construction, but by the late 1800s, many were built from copper or iron, or used iron hulls to protect the hull from fouling. As the cost of lumber and copper increased, iron and steel vessels became more popular.

3.2.6.20.1 Contaminants Associated with Shipbuilding

Initially, the seams in the hulls and decks of wooden steamships and ferry boats were caulked with coal tar pitch. However, because coal tar pitch was too brittle and scraped off, it was replaced by asphalt caulking pitch or "marine glue." The seams were first caulked with oakum (tarred fiber) and then filled with melted pitch. Marine glue also was used to waterproof the underside of wooden vessels (Webster et al. 1920). The hulls of wooden ships and boats were also treated with anti-fouling paints for which the active ingredient was typically finely divided metallic Cu. In addition, the paints used on wooden or iron and steel ships were pigmented with metallic compounds. For example, it was reported that a "red lead paint" could contain up to 80% of red Pb and was particularly effective as a base coat for metal surfaces exposed to marine conditions (Webster et al. 1920). Bituminous anti-corrosive coatings (also known as asphalt coatings) were used in ship bilges among other applications, to prevent deterioration of the ships' inner plating from corrosion. While touted as neither vulnerable to chipping or flaking by forming a permanently impervious coating, pumping of the bilges likely provided a mechanism for some portion of these materials to be discharged into the waterways. Anti-corrosive paints with metallic Zn, Zn oxide, shellac, alcohol, pine tar, and turpentine were applied to steel ships to insulate the metal in the anti-fouling coat (see below) from the steel plating, preventing corrosion and pitting. Anti-fouling paints applied to the hull and bottom of ships are designed to prevent marine growth adhering to the underwater surface of the hull. The active ingredients in these were typically mercuric oxide, copper oxide, and copper cyanide. Anti-fouling paints were designed to flake off to provide a new "active" surface (Webster et al. 1920). Therefore, contaminants associated with past shipbuilding activities primarily include hydrocarbons and PAHs from the use of coal tars and bituminous (asphalt) tars, as well as metals in paints, anti-corrosive paints and anti-fouling paints.

3.2.6.20.2 Shipbuilding near Newtown Creek

Shipbuilding began in Greenpoint in about 1832, when Neziah Bliss and Eliphalet Nott, two industrialists with a mutual interest in steam propulsion, established a shipyard there (Goodwin and Associates 2012; Silka 2006). At least a dozen shipbuilding firms followed Bliss and Nott's lead during the next two decades. The combined output of these Greenpoint yards was enormous and diverse, ranging from unrigged barges and scows, to full-rigged barges and sloops and a wide range of other gas and steam-powered vessel types (Goodwin and Associates 2012). During the 1840s through 1860s, hundreds of ships were built at shipyards, primarily along the navigable lower portion of the creek (Goodwin and Associates 2012; NYSDOT and FHWA 2005). Notable examples include the 360-foot-long wooden side-wheel steamship the *Great Republic* built in 1866, and the iron-hulled USS *Monitor*, built for the Union during the Civil War and launched from Greenpoint in 1862 (Goodwin and Associates 2012; Cimino [date unknown]). Ships were also launched from the beaches of the East River and included barges, sloops, clippers, and steamers. By 1870, very few shipbuilders remained.

3.2.6.21 Solid Waste Disposal and Landfilling

Land development activities, including dumping and filling, are documented near and along Newtown Creek since the beginning of the industrial development of the area in the late 1800s (Goodwin and Associates 2012). Historically, upland filling was used to fill historical marshland and creek channels, adjoin properties, and increase the current property footprint. Other filling activities were associated with channel improvements and restructuring. Historical fill materials may have included private and municipal trash, ash generated by burning trash, industrial waste, and dredge spoils from the creek (see Sections 4.1.1.2.4 and

4.1.1.4 of the draft DAR in Appendix J). In the 1800s and early 1900s, NYC filled marshes and swamplands with all types of municipal, commercial, and industrial wastes to create fastland from low-lying areas. In 1875, the Brooklyn Board of Health noted, “The gradual filling up of these marshes [Newtown Creek Marshes] and the numerous inlets extending into them, by the action of the tide, has been going on for many years” (City of Brooklyn 1876). Low-lying areas filled with wastes or ashes included:

- Oakland Street to Huron Street (City of Brooklyn 1876)
- Greenpoint Avenue and Norman Avenue (City of Brooklyn 1896)
- Greene Street to Greenpoint Avenue (City of Brooklyn 1896)
- Greenpoint Avenue, Newell Street, Norman Avenue, and Kingsland Avenues (NYCDOH 1898)
- Meeker Avenue and Grand Street near Varick Avenue (Emery 1895)
- Hunters Point (NYT 1874)

When NYC began to run out of areas to fill, “dumping boards” and “dumping grounds” were established where wastes were taken to the waterfront and dumped onto barges or scows for disposal (disposal locations are not known). In 1922, an internal USACE memorandum regarding policy and specifications for proposed dredging on Dutch Kills and Maspeth Creek included the following specification for disposal of dredged materials:

> Par. 34 Disposal of Excavated Material. – This paragraph is to follow the usual form, which provides for disposal of material at such localities as may be designated by the Supervisor of New York Harbor, or behind bulkheads in shoal waters or at the water’s edge, or the contractor may provide his own dumping ground subject to the approval of the contracting officer. (Rostock 1922)

By the 1960s, maintenance dredge spoils were typically combined with materials from other locations and disposed of in the New York Bight (USACE 1973).

NYC disposed of street sweepings, ash, offal, and other garbage in the salt marshes adjacent to the creek in the late 1800s and early 1900s (Goodwin and Associates 2012). Specifically, these activities occurred on the east side of Dutch Kills, between the LIRR tracks and Borden Avenue, along Whale Creek near the terminus of North Henry Street, and along the main stem at the terminus of Apollo Street (*Harper's Weekly* 1881; *Brooklyn Eagle* 1896; War Department 1916).

3.2.6.21.1 Contaminants Associated with Solid Waste Disposal and Landfilling Sites

Given the widespread dumping and landfilling activities that occurred in and around Newtown Creek beginning in the late 1800s and the types of waste products that were present in these wastes, it is likely that a number of contaminants were present in these wastes, including metals, petroleum products, PAHs, and PCBs (for waste disposed after 1929). It is also likely that these contaminants, many of which are the focus of the RI/FS for Newtown Creek, may have entered Newtown Creek through a number of transport pathways, although the exact nature and quantity of these contaminants that may have entered Newtown Creek over the past 125 years or so is not known.

3.2.6.21.2 Solid Waste Disposal and Landfilling Sites near Newtown Creek

The Whale Creek dumping ground noted in Section 3.2.6.21 was a free, public dump documented as early as circa 1890s, occupying approximately 15 blocks at the end of Humbolt Street at Whale Creek (see Sections 4.1.1.4 and 4.1.2.2 of the draft DAR in Appendix J; Goodwin and Associates 2012). A small ash dump was established circa 1920 east of Whale Creek, which evolved into a marine transfer station for municipal waste that operated until about 2001 (see Greenpoint incinerator under Section 3.2.6.9; NYSDOT and FHWA 2005). In May 1929, the Brooklyn Ash Removal Company was granted a permit by the War Department, United States Engineer Office, to build a dumping board and bulkhead located at the shore between Apollo and Van Dam streets (War Department 1929b). The 1932 building completion documents cite that construction was complete and that the additional permit conditions that "not more than one scow be moored alongside the bulkhead at all hours" were being complied with (War Department 1932).

3.2.6.22 Utilities

Throughout history, several operators provided various forms of power in the vicinity of Newtown Creek. Beginning in the 1820s, MGPs started to produce illuminating gas for use in municipal lighting (Moore Binder 1955; NYSDEC 2012a), and by the late 1800s, manufactured gas was a widely used fuel for heating and cooking. In the 1870s, electricity arrived as an alternative source of illumination, and in 1878, the Edison Electric Light Company was established. In 1881, the nation's first commercial electric power station was in operation in lower Manhattan. By 1982, all active electricity generators in NYC were operated by the Power Authority of the State of New York and Con Edison (NYCDEP 1983). Con Edison is now NYC's sole power company (St. James Press 1992). Con Edison operates several facilities, including Con Edison – 11th Street Conduit (DAR No. 110), a present-day electrical power distribution center located in Brooklyn (see Table 5-3, Figure 5-1, and Appendix B [specifically, the relevant upland site summary] of the draft DAR, as well as the new upland site summary, in Appendix J of this RI Report).

3.2.6.22.1 Contaminants Associated with Utilities

Historically, PCBs were widely used in transformers, capacitors, and oil-filled electrical equipment, which are an integral function of power distribution and utility operations (NYCDCP 1981). Pb was used in paper-insulated, Pb-covered cable. Historical documents report spills and leaks from electrical equipment and conduits (see the relevant upland site summary in Appendix B of the draft DAR in Appendix J of this RI Report). Hazardous substances such as asbestos, PCBs, and coal tar have been used or generated in the course of operations of Con Edison and its predecessors; they are present at sites and in facilities and equipment (Securities and Exchange Commission 2014). PCBs and other contaminants may have entered Newtown Creek through a number of transport pathways, including direct discharges through stormwater sewers or overland transport.

3.2.6.22.2 Utilities Sites Near Newtown Creek

Con Edison operates at least 1,270 facilities (e.g., subsurface manholes, vaults, and service boxes, which are interconnected by underground conduits that house electrical distribution cables) within a 1-mile radius of Newtown Creek (see Table 5-3 and Figure 5-1 of the draft DAR in Appendix J). In addition to their underground distribution network, they currently operate or have operated in the past the following three sites that are either creek-side or near the creek. Trace or higher amounts of PCB oil have been listed in manifests at these sites located within the Newtown Creek sewershed:

- **The Con Edison – 11th Street Conduit (DAR No. 110), (see Table 5-3, Figure 5-1, and Appendix B [specifically, the relevant upland site summary] of the draft DAR, as well as the new upland site summary, in Appendix J of this RI Report)**. Located in Brooklyn, the Con Edison 11th Street Conduit has operated a conduit tunnel from 1964 to the present day to house underground transmission lines that connect the Rainey and Farragut electrical substations. The conduit passes under Newtown Creek and contains an electrical distribution cable that extends between Brooklyn and Queens. Since the electrical distribution cables within the conduit are oil-filled, an oil-water separator was installed to separate any oil from collected waters prior to discharge. The site was also used as a coal yard circa the 1920s. Contaminants commonly associated with coal yards and electrical power distribution include PCBs, TPH compounds, PAHs, and VOCs. Sources of these contaminants include coal, equipment and vehicles used in coal yard operations, electrical distribution equipment, and spills. Coal-storage contamination is discussed in detail in Section 3.2.6.5. In addition, direct discharge of stormwater and wastewater is a potentially complete historical pathway and a complete current pathway. Stormwater at the site is expected to infiltrate into the ground, flow overland toward Newtown Creek, or discharge to the creek via a local storm drain system (potentially at Outfalls NCB-246 and NCB-248). Seepage of groundwater through the conduit walls and infiltration of rainwater through the riser shafts located at Ash Street and at 11th Street and 47th Road is discharged to Newtown Creek at Outfall NCB006131-002 (Con Edison 1994).
- **The Con Edison – Maspeth Substation (DAR No. 4) (see Table 5-3, Figure 5-1, and Appendix B [specifically, the relevant upland site summary] of the draft DAR in Appendix J of this RI Report)**. The Con Edison Maspeth Substation, located on Rust Street between Maspeth Avenue and Grand Avenue, operated as an electrical substation from 1922 to 1985. Con Edison sold the site in 1996. Tire-recapping operations occurred at the site for 2 years, and textile storage operations have occupied the site from 1999 to the present day. Contaminants found at the site include dielectric oil (containing PCBs), VOCs, and SVOCs. Sources of these contaminants include discharges, leaks, and spills originating from the equipment and other unknown sources at the site. There are six documented NYSDEC spills at the site from 1997 to 2005, involving petroleum, hazardous material, hydraulic oil, and transformer oil. Based on the presence of PCBs in the former transformer yard, the site is listed on the New York

State Environmental Site Remediation Database as a "Class A" Voluntary Cleanup Program site (i.e., a non-registry site in any remedial program where work is underway and not yet complete). Investigations conducted at the site since 1989 revealed the presence of petroleum (transformer oil) contaminated with PCBs in both soil and groundwater. Following excavation and remedial actions at the site, subsurface soil samples collected between 1996 and 2001 still contained PCBs at concentrations greater than the corresponding NYSDEC Recommended Soil Cleanup Objectives, in several soil borings to depths of at least 17 feet below ground surface. Light nonaqueous phase liquid (LNAPL) containing PCBs was detected during RI activities in several groundwater monitoring wells at approximately 15 to 17 feet below ground surface; the maximum PCB concentration detected in the NAPL was 328 parts per million, consisting entirely of Aroclor 1260. The area of LNAPL was excavated, and subsequent groundwater monitoring over a 2-year period has intermittently detected LNAPL (up to 0.4 foot thick) in some on-site and off-site wells.

- **The Con Edison – Newtown Substation (DAR No. 5) (see Table 5-3, Figure 5-1, and Appendix B [specifically, the relevant upland site summary] of the draft DAR in Appendix J of this RI Report)**. The Con Edison Newtown Substation located in Long Island City was purchased by Con Edison in 2006 and has operated as an electrical substation from 2010 to the present day. The site was also used for auto parts storage circa 1947 and for truck leasing from circa 1970 to 2006. Historically, the site was a Resource Conservation and Recovery Act large quantity generator, and available hazardous waste manifests indicate the facility shipped non-listed ignitable wastes (D001), corrosive hazardous wastes (D002), Pb (D008), Hg (D009), and benzene (D018). Thirteen USTs containing "other" product or gasoline were closed prior to 1991. There are 10 documented NYSDEC spills at the site from 1989 to 2010, involving gasoline, No. 2 fuel oil, petroleum, hydraulic oil, and dielectric fluid. No soil or groundwater investigations at the site were identified in documents available for review. Uniform Hazardous Waste Manifest forms document that drums containing unspecified benzene contaminated liquids (i.e., oil and water), solids, and unspecified Pb debris were removed from the site between 2007 and 2011.

3.2.6.23 Waste Oil Refining Operations

Waste oil refining operations comprise processes that simply remove water and insoluble components from waste oils, so that the oils can be burned as a heat source, as well as processes that re-refine oils to be used again for automotive purposes.

3.2.6.23.1 Contaminants Associated with Waste Oil Refining Operations

Contaminants associated with waste oil refining operations are similar to petroleum refining and bulk storage operations (see Section 3.2.6.15). However, waste oil refining or recycling operations may also process oils with additional contaminants, including PCBs, PAHs and metals, depending on the source of these waste oils.

3.2.6.23.2 Waste Oil Refining Operation Sites near Newtown Creek

Several waste oil refining operations have operated along the creek, including Quanta/RAD II (DAR No. 39), located in Queens; and B.C.F. Oil (DAR No. 27), located in Brooklyn along the banks of English Kills (see Table 5-3, Figure 5-1, and Sections 7.7 and 7.8, respectively, of the draft DAR in Appendix J).

The 1.9-acre B.C.F. Oil (DAR No. 27) site is now a vehicle impoundment lot. In the late 1800s, the majority of the site was an embayment (War Department 1884; see the relevant upland site summary in Appendix B of the draft DAR, as well as the new upland site summary, in Appendix J of this RI Report). The embayment was filled to the present-day shoreline by the early 1930s (Sanborn 1933; Malcolm Pirnie 2010). Throughout the next four decades, from 1933 to 1979, the site was in use as a bulk oil terminal. In 1979, the site was modified for waste oil recycling. From 1980 to 1994, Calleia Brothers, Inc., and B.C.F. Oil used the site for waste oil refining (see Section 3.2.6.15 for additional information regarding the use of the site for waste oil refining). The facility was closed in 1994, after PCB-contaminated oil was discovered (Malcolm Pirnie 2010). The RI conducted by Malcolm Pirnie reported that the sporadic low-level detection of PCBs may be attributable to historical fill materials, because much of the site was created from pavement rubble imported during the 1970s to expand the usable area of the site (Malcolm Pirnie 2010; Rust Environment and Infrastructure 1998). Waste oil loading/unloading areas at the site were surrounded by secondary containment dikes. Stormwater in this area was conveyed to an oil-water separator for treatment prior to discharge. In August 1994, NYSDEC approved an emergency authorization to discharge wastewater from the site to English Kills, because heavy precipitation overwhelmed the oil-water separator. NYSDEC contracted to have a bypass installed to allow stormwater to flow through a direct pipe into English Kills, bypassing the oil-water separator (NYSDEC 1994a, 1994b). The authorization indicated that the pollutants potentially released in the discharge would include oil; BTEX; and PCBs. In addition, beginning in May 2000, USEPA conducted an emergency response action to address concerns about possible leakage from unmaintained USTs, ASTs, and drums. By October 2001, USEPA had removed more than 800,000 gallons of PCB-contaminated oil, wastewater, and sludge, and cleaned and closed in place the ASTs and USTs on site.

The Quanta/RAD II (DAR No. 39) site has a long history of industrial use (see the relevant upland site summary in Appendix B of the draft DAR in Appendix J of this RI Report). By the 1930s, the Triplex Oil Company—a crankcase oil refinery—occupied the site (CCBQ 1936, 1941). Although ownership of the business and property changed hands, the site featured similar uses (e.g., oil recycling, disposal, and refining) until the early 1980s (Golder 2005). Between 1972 and 1980, Russell W. Mahler created six different corporations that operated at the site, including Quanta Resources Corporation, which initiated bankruptcy proceedings on October 6, 1981. Concern for public welfare prompted NYCDEP officials to execute an emergency remedial action and enter the property to assess and respond to unsafe conditions (EDR 2010; Woodward-Clyde 1984). By the end of 1982, more than 500,000 gallons of liquids were removed from the site, including recyclable oil, PCB oil, PCB sludge, cyanide solution, and PCB-contaminated diesel fuel. Hazardous wastes generated at the site included As, Cr, Pb, benzene, VOCs, SVOCs, and PCBs. The site was accepted into the Brownfield Cleanup Program in 2006. The Record of Decision was signed on February 9, 2007. Site redevelopment was in tandem with RAD I (DAR No. 41; formerly Van Iderstine Company). The RAD I site includes two discontiguous tax lots (known as the North and South Capasso properties). The North Capasso property (Lot 41) was adjacent to the north/northeast site property boundary. The South Capasso property (Lots 279 and 280) was located to the south/southwest of the site on the far side of the LIRR tracks adjacent to Newtown Creek (EDR 2010). Both sites (RAD I and Quanta/RAD II) have been through RIs where NAPL, PCB fluids, and other contamination have been found at these sites and an adjoining site. Environmental covenants have been filed restricting access to site soils and groundwater.

3.2.7 Current Upland Activities, Uses, and Marine Facilities

Today, the predominant land use around Newtown Creek and the tributaries remains industrial, with pockets of mixed use, commercial, and residential developments. Industries located adjacent to the creek in 2014 are shown in Figures 3-19a through 3-19j. Typical activities include the following: warehouse and distribution facilities; vehicle storage and maintenance; electrical distribution; plastics and foil manufacturing; waste transfer yards and recycling facilities; road service support facilities; construction materials storage; facilities that store electrical equipment; scrap metal processing facilities; lumberyards; ready-mix concrete plants; bulk fuel distribution terminals; railroads (e.g., tracks, yards); utilities; and municipal wastewater treatment (see Sections 4.1.2.5, 5.6, and 7 of the draft DAR in Appendix J). Since developing the draft DAR in 2012, and the draft SSAM (Anchor QEA 2014m), nine new upland site summaries have been developed, and minor edits have been made to six others. These provide updates on the status of the site investigations and include information on more recent upland remedial activities, and they are included in Appendix J. The nine new upland site summaries are for the following:

- 8 Rewe Street, LLC
- Bayside Fuel Oil Depot
- B.C.F. Oil
- Borden Realty Co.
- Charles J. King, Inc.
- Con Edison – 11th Street Conduit
- Long Island Rail Road – Johnson and Morgan Avenue
- Review Avenue Development I
- Western Beef Properties

The six upland site summaries for which minor edits have been made are the following:

- Buckeye Pipeline Facility
- Cipico Construction, Inc.
- Frito Lay and Frito Lay II
- Hugo Neu
- Manhattan Poly Bag
- Morgan Oil

There were nine active marine facilities in operation along the waterway in 2014. Similar to historical creek uses, these facilities fall into petroleum handling or scrap metal hauling categories. Active marine facilities are listed within the creek-side site uses in Figures 3-19a through 3-19j. Newtown Creek is located within the New York State Coastal Zone. Within the Coastal Zone designation, Newtown Creek is designated one of six NYC SMIAs (NCBOA 2012). SMIA designations were first developed by NYC in the 1992 Comprehensive Waterfront Plan (NYC 2011). These designations, typically characterized by clusters of industrial firms and water-dependent businesses, were intended to protect and encourage concentrated working waterfront uses. The Newtown Creek SMIA is the largest of the six SMIAs in NYC, encompassing approximately 780 acres (NYC 2011).

Public access from land to the Study Area is restricted in many areas by physical controls (e.g., fences) and security/surveillance controls operated by industrial facilities. These access restrictions to the Study Area significantly limit the public's ability to engage in recreational activities in Newtown Creek, including opportunities to fish and crab from the shoreline within the Study Area. Notwithstanding these access restrictions, there are some defined public access areas to the shoreline of Newtown Creek with some facilities (e.g., maintained parks, kayak and canoe launch areas, and a nature walk) as well as other areas with no facilities that allow for informal public access to the Study Area shoreline (e.g., unmaintained end of road access points and access from bridges). Facilities are defined as any development that is present to enhance shoreline recreational activities, such as pedestrian/bike paths, athletic facilities (e.g., basketball courts), benches or platforms for resting, and canoe/kayak launches. Depending on the type of access and whether facilities are available, recreational activities available at these locations can include shoreline-based recreational activities such as walking, jogging, sitting, biking, bird/wildlife viewing, and attending classes and other educational activities. In-water recreational activities can also include boating (canoe/kayak), angling/crabbing, and swimming activities where contact with the surface water and sediments in the Study Area would be more likely. Swimming, however, is considered an extremely rare activity in the Study Area, as the final public health assessment for Newtown Creek states, "NYSDOH does not have clear evidence that swimming in Newtown Creek is occurring" (NYSDOH 2014). The public health assessment conducted by NYSDOH was driven by concerns regarding exposure to waterborne pathogens associated with raw sewage discharges from CSOs during rain events. In addition, USEPA has observed swimming in only two locations, Manhattan Avenue Park (where swimming is prohibited) and at the location of an illegally moored boat in English Kills. In addition to the existing formal and informal access points, future development of additional formal and informal recreation access areas on Newtown Creek may occur (e.g., Box Street Park, Vernon Street End Area, and North Henry Street Area [Newtown Creek CAG 2021]), although the scope and timing of any potential future development is not known.

In addition to recreational activities, the shoreline of Newtown Creek is used for education activities that include field trips, classes along the shoreline, and assisting research projects that involve collection of samples along Newtown Creek shorelines. These educational activities generally occur at the formal access areas but may also occur at privately owned properties where owners allow individuals and groups access to the shoreline of Newtown Creek for educational purposes.

The following is a list of existing formal and informal public access areas with a brief description of each area and the types of recreational activities available at the location:

- **Hunter's Point South Park:** This 11-acre waterfront park is located at the confluence of the East River and Newtown Creek in the Long Island City neighborhood of Queens. This park includes facilities to enhance shoreline recreational activities, including playgrounds, a dog run, a bikeway, a waterside promenade, picnic terraces, athletic facilities, a 30-foot-tall platform for viewing the skyline and waterfront, and a kayak and canoe launch area. This park allows for shoreline recreational activities as well as opportunities for canoe and kayak recreation and potential angling/crabbing.
- **Manhattan Avenue Park:** This park consists of a small parking lot located at the north end of Manhattan Avenue in the Greenpoint neighborhood of Brooklyn. The park includes approximately 184 feet of vertical concrete bulkheads with metal railings along the creek that are used to launch canoes and kayaks. Park rules prohibit swimming, diving, or bathing. This area is suitable for shoreline recreational activities, canoe and kayak recreation, and potential angling/crabbing.
- **Under the K Bridge Park:** This 7-acre park includes pedestrian and bike paths and is located under the Kosciuszko Bridge. This park is suitable for shoreline recreational activities and potentially canoe and kayak recreation and angling/crabbing.

- **Newtown Creek Nature Walk:** This nature walk consists of a path along the perimeter of the Newtown Creek WWTP in the Greenpoint neighborhood of Brooklyn. Swimming, diving, canoe/kayak launching, and fishing are expressly prohibited. The Newtown Creek Nature Walk allows for shoreline recreational activities.
- **North Brooklyn Boat Club Canoe/Kayak Launch:** The North Brooklyn Boat Club's boatyard provides a canoe/kayak launching area on the west side of the Pulaski Bridge in Brooklyn. This current location is considered a temporary location while a permanent boathouse location for the Boat Club is found and developed. This area is suitable for canoe and kayak recreation and education activities.
- **Plank Road Area:** This access area is an informal end-of-road public access area to the Newtown Creek waterfront at the terminus of 58th Road in the Maspeth neighborhood of Queens. This area is suitable for shoreline recreational activities.
- **Metropolitan Avenue Bridge:** This access area provides an informal bridge access to the English Kills tributary to Newtown Creek in Brooklyn. Although there are no facilities at this location, this area is suitable for shoreline recreation and potential canoe and kayak recreation and angling/crabbing.
- **Borden Avenue Street End on Dutch Kill Area:** This informal street end and bridge access is located on the Dutch Kills tributary to Newtown Creek in Queens. Although there are no facilities at this location, this area is suitable for shoreline recreation and potential boat/kayak recreation.
- **Apollo Street End Area:** This informal street end access is located in the Greenpoint neighborhood of Brooklyn. Although there are no facilities at this location, this area is suitable for shoreline recreation.
- **Maspeth Avenue Street End Area:** This informal street end access is located in the East Williamsburg neighborhood in Brooklyn. Although there are no facilities at this location, this area is suitable for shoreline recreation.

3.2.8 Discharges to Newtown Creek

As discussed in the preceding subsections of Section 3, many decades of urban development along the creek and in the surrounding upland areas have altered the hydrology of the watershed draining to Newtown Creek. The tributaries and adjacent marsh areas were filled, and large portions of the surrounding neighborhoods were covered with impervious surfaces (i.e., buildings, pavement). Over time, tributary flows to Newtown Creek were eliminated and replaced by point source and overland flow discharges.

In the 1800s and early 1900s, untreated stormwater, sewage, and industrial wastewater were generally discharged directly to Newtown Creek. Although the present-day system of intercepting sewers (i.e., interceptors) were conveying flows to the Bowery Bay WWTP and Newtown Creek WWTP by 1967, direct discharges of stormwater and CSOs to the creek continued and are ongoing today. The deleterious effects of point source discharges on water quality in Newtown Creek were noted by public health officials, engineers, and others as early as the late 1800s (City of Brooklyn 1897; Metropolitan Sewerage Commission 1910; Hazen and Sawyer 1960).

Interceptor sewers were constructed in the early 1900s to convey flows in Brooklyn and Queens to the East River, although sewer flow continued to discharge to Newtown Creek via relief outlets and storm overflows (Metropolitan Sewerage Commission 1910). The Newtown Creek WWTP, Bowery Bay WWTP and the present-day system of intercepting sewers had been constructed by 1967. CSOs, consisting of wastewater (e.g., sanitary sewage and industrial process water) and stormwater, continue to discharge to Newtown Creek when the capacity of the WWTP infrastructure is exceeded (Lang et al. 1974; NYCDEP 2011a). Stormwater discharges directly to the creek in areas served by MS4s. Regulation of private and municipal point source discharges into Newtown Creek increased with the 1972 passage of the CWA and subsequent amendments, which established the National Pollutant Discharge Elimination System (NPDES) and State Pollutant Discharge Elimination System (SPDES) programs (NYCDEP 2011a). However, direct discharges to Newtown Creek are ongoing today. An overview of current and historical CSO discharges and regulation of CSO discharges to Newtown Creek is presented in this section and shown on the timeline in Figure 3-7. An LTCP for CSO discharges to Newtown Creek was approved on June 27, 2018 (NYSDEC 2018a).

Current point source discharges to the Study Area are individually permitted discharges, WWTP treated effluent overflows,[36] CSOs, and stormwater (including overland flow). Individually permitted discharges include treated effluent[37] from groundwater remediation and dewatering systems, stormwater discharge from industrial sites, hydrostatic test water,[38] and discharge from secondary containment. Stormwater discharges (including overland flow) are runoff from private, commercial, and industrial sites—and from roads and open spaces.

3.2.8.1 Historical Discharges to Newtown Creek

In the 1800s and early 1900s, direct discharge to Newtown Creek was the primary method for disposal of local stormwater, sewage, and industrial wastewater. For example, investigations conducted by the Brooklyn and Long Island City Boards of Health documented discharges to the creek of waste liquor from grease vats and digesters at fertilizer and fat rendering facilities, oily wash water generated during the kerosene treating process at refineries, and condensate from exhaust hoods at varnish and printing works (BPL 1890; NYSL 1900; Baker and Kent 1887). The public and the regulators (local and state boards of health) focused on minimizing nuisance odors, which were believed to transmit disease, and urged manufacturers to employ processes that transferred contaminants from air to water (essentially scrubbing emissions). The effluent from these air-scrubbing systems was discharged to Newtown Creek (Hurley 1994).

Municipal sewers were constructed in many areas prior to the consolidation of the boroughs of Brooklyn and Queens into NYC, which occurred in 1898. In 1857, the legislature authorized the Board of Commissioners to design and construct a system of sewerage for the City of Brooklyn. Prior to 1857, there were 5.5 miles of sewer in Brooklyn, constructed

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[36] With regard to "treated effluent overflows," the Newtown Creek WWTP began operating in 1967 and employed a modified aeration treatment process. Upgrades to achieve secondary treatment were completed in 2011. Although the treated effluent from the Newtown Creek WWTP is normally discharged outside the Study Area to the East River, under certain high-flow conditions (which are described in Section 3.2.8), treated effluent from the Newtown Creek WWTP is discharged inside the Study Area via the high-relief outfall (NCB-002) to Whale Creek (Cunetta and Feuer 1968; Greeley and Hansen et al. 2010; NYCDEP 2011d, 2017).

[37] With regard to "treated groundwater effluent," the degree of treatment is variable and documented in each individual SPDES permit.

[38] Hydrostatic test water is water placed in pipelines, tanks, or other vessels and raised to greater than atmospheric pressure to confirm the integrity of the pipeline, tank, or vessel.

primarily to drain low-lying areas (Board of Water Commissioners 1867). By 1895, Brooklyn's sewer system included more than 500 miles of sewer lines (NYT 1895).

Early sewers typically discharged directly to the nearest waterbody without treatment. In 1897, the Brooklyn Department of City Works noted in the annual report to the Brooklyn Common Council that sewage from municipal sewers in Huron Street, Humbolt Street, Norman Avenue, and other nearby streets were discharged to "low lying lands about the lines of old Whale Creek" and were filling the area "obstructing more and more the original course of drainage" (City of Brooklyn 1897). The same year, the Brooklyn Department of Health noted that "the city has not yet begun the construction of the intercepting sewers, designed last year, to carry all sewage to the East River, instead of allowing it to flow into Newtown Creek. Until this is done, we cannot hope for much improvement in the condition of the creek" (City of Brooklyn 1897).

In November 1911, the Metropolitan Sewerage Commission collected water samples from four locations in "densely polluted" Newtown Creek and analyzed the samples for DO content. Samples were collected from 1 and 20 feet below the surface at the mouth of the creek. Reported concentrations were highest at the mouth of the creek and progressively decreased in samples taken farther from the East River—3.41 mg/L at the mouth of the creek, 0.84 mg/L at Vernon Avenue Bridge, and 0.29 mg/L at Greenpoint Avenue Bridge, and no detectable DO was present in the sample collected at Meeker Avenue Bridge (Metropolitan Sewerage Commission 1912).

The Metropolitan Sewerage Commission reported that Newtown Creek was in a "very objectionable condition" as a result of "a considerable quantity of manufacturing wastes and the flow of a few sewers." Intercepting sewers had been constructed by 1910 in some areas near the creek to convey flows to the East River. However, direct discharges continued in areas without interceptors, and untreated discharges to Newtown Creek via relief outfalls (i.e., CSO outfalls) continued to occur during wet weather events when the capacity of the interceptors was exceeded. The Commission's report noted that sewage and wastes discharged into Newtown Creek were often not carried away, because the movement of water was only due to the rise and the fall of the tides, and there was no circulation

(Metropolitan Sewerage Commission 1910). Relief outlets documented by the commission are described in the following paragraphs and are shown in Figure 3-20.

In Queens, an intercepting sewer had been constructed under the LIRR tracks to convey sewage from portions of Long Island City to the East River. Overflows from this interceptor discharged to Newtown Creek via a 192-inch-diameter outfall at Seneca Avenue (near present-day NCB-083); 24-inch-diameter outfalls at Vernon Avenue (present-day BB-014), Greenpoint Street (present-day BB-011), and Pearsall Street (present-day BB-012); and a 50.5-inch-by-39-inch outfall at East Avenue (present-day BB-013) (see Figure 3-20; Metropolitan Sewerage Commission 1910).

The Brooklyn-Queens Interborough Sewer conveyed sewage from Queens and Brooklyn to the East River, discharging near the Williamsburg Bridge. Overflows from the Interborough Sewer discharged to Newtown Creek at several locations, including through a 180-inch-diameter outfall at the head of East Branch (present-day NCB-083). Relief outfalls (i.e., CSO outfalls) discharging to Newtown Creek also existed at Oakland Avenue (36-inch-diameter; present-day CSO Outfall NCB-021) and at Pink Street (18-inch-diameter) (see Figure 3-20; Metropolitan Sewerage Commission 1910).

In 1927, the Brooklyn Bureau of Sewers acquired land on Johnson Avenue for the purpose of constructing and maintaining a stormwater sewer outlet at the head of English Kills (present-day NCB-015) (Bureau of Sewers 1927).

In 1939, more than 20 years after the intercepting sewers had been constructed, the Chief of Engineers, U.S. Army, sent letters of complaint to the Greenpoint Civic Council and the North Atlantic Division Engineer regarding the pollution in Newtown Creek. According to these letters, Newtown Creek was “black in color, septic even in winter and gives off bubbles of hydrogen sulfide gas at all times” (note that such bubbles contribute to gas ebullition) and is “generally considered as the most badly polluted locality within the New York Metropolitan district.” The Chief of Engineers wrote that “in addition to industrial wastes, the creek receives ever increasing amounts of sewage.” A list of known sewers that discharged to Newtown Creek, including 22 NYC-owned outfalls (13 combined outfalls ranging from 12 to 66 inches in diameter; six storm [i.e., relief outlets, CSO] outfalls ranging from 20 to 186 inches in diameter; and three sanitary outfalls ranging from 12 to 60 inches in diameter) and five industrial outfalls (fat reduction for the Van Iderstine Company; Long Island Soap Company; Fischer Bros.; Joseph Rosenberg's Sons; and Geiger Products Company) was provided.[39] The locations of outfalls identified in this letter are shown in Figure 3-21 (War Department 1939; USACE and USN 1939).

By 1958, the Long Island City interceptor had been constructed and was conveying combined flows from areas surrounding Dutch Kills to the Bowery Bay WWTP, located north of the Study Area in Astoria. The Bowery Bay WWTP opened in 1939, with a design capacity of 40 million gallons per day (MGD) and employing an activated sludge treatment process. The plant was upgraded to achieve secondary treatment in the late 1970s (NYCDEP 2017). Flows of stormwater, sewage, and industrial wastewater from Long Island City continued to discharge to Newtown Creek until the late 1950s, when the Long Island City interceptor was constructed, and the Bowery Bay plant capacity was expanded to 120 MGD (NYCDPW 1952, 1954, 1956, 1958; Lang et al. 1974).

In 1960, the NYSDOH Water Pollution Control Board published a report and surveys on studies of the Lower East River and tributaries, including Newtown Creek. The purpose of the report was to compile information, including "hydrology, land use, present and future water uses, present sources of pollution and the extent of water defilement, and the effect of pollution upon the water and land uses" for consideration in the development of waterbody classifications (Hazen and Sawyer 1960).

The NYSDOH study included documentation of municipal and industrial discharges to Newtown Creek observed during field work in 1959 and 1960. Ten outlets (i.e., CSO outfalls) corresponding to present-day CSO Outfalls BB-011, BB-012, BB-004, BB-026, BB-040, BB-042, BB-043, BB-013, BB-014, and BB-015 were documented and noted to be intercepted by the Bowery Bay interceptor (also known as the Long Island City interceptor). Five outlets corresponding to present-day CSO Outfalls NCB-083, NCB-015, NCB-022,

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[39] Historical outfall types differ from present day terminology. Storm outfalls were relief outfalls from the combined system that discharged during wet weather events, similar to present-day CSOs. Sanitary outfalls discharged sewage directly to the creek. Combined outfalls discharged sewage and stormwater; however, unlike present-day CSOs, these were not overflows, as there was no treatment system for the combined sewers.

NCB-023, and NCB-024 were documented and noted to be not intercepted in 1960, pending completion of the proposed Newtown Creek interceptor (also known as the Morgan Avenue interceptor). Locations of the outfalls documented in this report are provided in Figure 3-22 (Hazen and Sawyer 1960).

More than 50 industrial discharges to Newtown Creek were also described in the NYSDOH report (see Table 3-2 and Figure 3-22). These discharges are located throughout the Study Area, in the main stem of the creek and the tributaries. According to the report, the majority of the industrial discharges were untreated; however, at some facilities, stormwater and wastewater flows were routed through oil-water separators and settling basins prior to discharge. Several of the industrial outfalls noted 20 years earlier by the Chief of Engineers were also identified in the NYSDOH report (Hazen and Sawyer 1960). These facilities (and associated DAR numbers as applicable) are as follows:

- RAD I (DAR No. 41): Van Iderstine Company (a facility that processed animal byproducts) was noted to discharge drippings and equipment washup water from settling basins via a submerged 18-inch-diameter outfall. Scum and floating solids on the surface and some gasification from bottom deposits were observed near that outfall.
- Rencoa, Inc. (DAR No. 139):
  - Joseph Rosenberg's Sons (a tallow manufacturer), routed equipment washup and floor drainage through a grease trap prior to discharge to the creek via a 12-inch outfall.
  - Long Island Soap Company (which had discharged directly to the creek in 1939) discharged water from heating coils, truck washing, and surface drainage to the city sewer.
- Pinkas Fischer (DAR No. 136): Pinkas Fischer Co. Inc., (formerly Fischer Bros.) employed a six-compartment grease trap as pretreatment for process wastes from their rendering facility prior to discharge via a 12-inch submerged outfall.

Additional industrial discharges documented in Table 4 of the NYSDOH study (Hazen and Sawyer 1960) included the following (also see Table 3-2):

- Brown, turbid, steaming effluent was discharged by National Sugar Refinery Co., from two submerged 20-inch-diameter outfalls. The report noted that an area of

50 feet by 100 feet was impacted by this discharge. Discharges from the facility also occurred via submerged 36-inch-diameter and 6-inch-diameter outfalls.

- Former Laurel Hill Site (DAR No. 16): The PDRC copper refinery discharged stormwater and wastewater to the creek via 10 outfalls, which ranged in size from a few inches to a few feet in diameter. Water discharged from slag granulating at this facility passed through a sedimentation pit prior to discharge.
- Greenpoint Energy Center (DAR No. 32): Brooklyn Union Gas, which was operating intermittently, routed process water from condensers and oil strippers through separators and coke filters prior to discharge.
- Several bulk fuel oil and bulk petroleum handling facilities, including Shell Oil Co.,[40] Mobil Oil Co.,[41] Paragon Oil Co.,[42] Gulf Oil Corp.,[43] Preferred Oil Co.,[44] Premium Coal and Oil,[45] Texaco Inc.,[46] and Sinclair Refining Co.,[47] routed stormwater from areas used for bulk handling of petroleum products to separators prior to discharge. However, other petroleum facilities, including Esso Standard Oil,[48] Metropolitan Petroleum Corp.,[49] Morania Oil Co. Inc.,[50] Sun Oil Co.,[51] and Amoco[52] discharged untreated stormwater to Newtown Creek. Triplex Oil Refinery[53] was noted to discharge untreated stormwater and sanitary wastes to Newtown Creek via a 4-inch-diameter outfall; effluent from this outfall was described as black (Hazen and Sawyer 1960; also see Table 3-2).

Fourteen of the industrial discharges documented by NYSDOH were discharged to a municipal sewer that was not intercepted and subsequently discharged to Newtown Creek without treatment. For example, Leviton Mfg. Co., Inc. (electrical wiring devices),

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[40] Located on Motiva Brooklyn Terminal (DAR No. 50)

[41] Located on ExxonMobil Greenpoint Remediation Project (DAR No. 53)

[42] Located on Empire Merchants/Former Paragon Oil Terminal (DAR No. 200)

[43] Located on Malu Properties/Former Ditmas Oil/Former Gulf Oil (DAR No 123)

[44] Located on Manhattan Polybag (DAR No. 130)

[45] Located on Morgan Oil (DAR No. 60)

[46] Not located on a DAR site

[47] Located on New York Paving, Inc. (DAR No. 214)

[48] Located on Newtown Creek WWTP (DAR No. 11a)

[49] Located on ExxonMobil Greenpoint Remediation Project (DAR No. 53)

[50] Located on B.C.F. Oil (DAR No. 27)

[51] Not located on a DAR site

[52] Not located on a DAR site

[53] Located on Quanta/RAD II (DAR No. 39)

discharged cooling water, plating wastes, and plant clean-up water to a non-intercepted city sewer (Hazen and Sawyer 1960).

As part of the field survey, on September 24 and September 30, 1959, surface water samples were collected at four locations in Newtown Creek (Pulaski Bridge, Borden Avenue Bridge, Greenpoint Bridge, and Grand Avenue Bridge) and analyzed for water quality parameters. DO was not detected in all eight samples. Five-day biochemical oxygen demand ranged from 12 to 39 parts per million, and coliform density ranged from 910,000 to 9,300,000 per 100 milliliters (mL) (measured as Most Probable Number [MPN]; Hazen and Sawyer 1960).

The NYSDOH report concluded that Newtown Creek was “grossly polluted,” and the best usage of the creek was wastewater disposal. The report stated that “overshadowing all other factors in determining water quality in the lower East River will be the operation of the proposed Newtown Creek Sewage Treatment Plant. When in operation, the plant should reduce the organic load to the lower river by approximately 75%” (Hazen and Sawyer 1960).

The Newtown Creek WWTP began operating in 1967, initially receiving flows from Queens and Brooklyn. In 1968, flows from Manhattan were conveyed to the plant through a force main under the East River. The Newtown Creek WWTP was designed to treat an average of 310 MGD, using a modified aeration process (Lang et al. 1974). The design of the plant included a spillway to discharge effluent to Whale Creek when the capacity of the plant was exceeded. During emergency operations, the entire flow could be discharged to Whale Creek via the spillway (Cunetta and Feuer 1968). Section 301 of the CWA established a required performance level, referred to as “secondary treatment,” which all publicly owned treatment works were required to meet by July 1, 1977. The secondary treatment process includes physical, chemical, and biological processes that remove at least 85% of carbonaceous biochemical oxygen demand – 5 day and TSS. As part of a consent judgment with NYSDEC, the Newtown Creek WWTP underwent a $5-billion extensive upgrade to achieve secondary treatment and to increase the plant’s wet weather treatment capacity from a minimum of 620 MGD to a minimum of 700 MGD. NYCDEP began the upgrade in 2000, and the plant achieved secondary treatment in June 2011 (NYCDEP 2011a).

3.2.8.2 Current Discharges to Newtown Creek

The Newtown Creek drainage area comprises approximately 7,300 acres in Brooklyn and Queens. Approximately 66% of this area is served by combined municipal sewer infrastructure. The remaining area is primarily served by municipal separate sewage and stormwater systems[54] (see Figure 3-5). In some areas near the creek, stormwater is discharged to the creek via privately owned infrastructure (NYCDEP 2011a). More than 300 private and municipal outfalls have been documented along Newtown Creek and its tributaries, some of which may be abandoned or no longer in use. These outfalls are shown in Figures 3-19a through 3-19j, and the point source inventory is discussed in detail in Section 2.1 of Appendix E and shown in Figures E2-4a through E2-4j. Discharges to the Study Area are CSOs, WWTP treated effluent overflow, stormwater (including overland flow and individually permitted stormwater discharges), and individually permitted treated effluent discharges from groundwater remediation and dewatering systems (see Figure 3-23). Solids and chemical loads that enter the Study Area from point source discharges and overland flow represent a potentially significant ongoing source to Newtown Creek. Section 5.1 presents estimated current loads based on concentration data collected during the Phase 2 point sources sampling program and flows derived from the 2015 geo-neutral point source version of the NYCDEP InfoWorks model, data extracted from published reports, and/or based on site-specific data provided by owners of some discharges.

3.2.8.2.1 Combined Sewer Overflows

In portions of the Newtown Creek drainage area served by municipal combined sewer systems, stormwater and sewer discharges enter the same pipe. During typical conditions, combined flows are conveyed to either the Newtown Creek WWTP or the Bowery Bay WWTP. When the capacity of the combined infrastructure is exceeded, combined flows (i.e., sewage and stormwater) are discharged directly to Newtown Creek through the 22 CSO outfalls shown in Figure 3-23. The locations of interceptor sewers (large pipes designed to intercept and redirect combined flows from discharging directly into surface waterbodies) within the Newtown Creek drainage area are also shown in Figure 3-5. The SPDES permits for the Newtown Creek WWTP (Permit No. NY0026204) and Bowery Bay WWTP (Permit No. NY0026158) include requirements for managing CSOs (NYCDEP 2011a). Gross

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[54] MS4s discharge directly to Newtown Creek.

floatable debris also flows from the CSOs. NYCDEP has an ongoing program to remove gross floatable debris, by collecting it at booms located at the head of the tributaries, downstream of the CSOs. These booms also restrict the navigability of the tributaries.

3.2.8.2.2 Newtown Creek WWTP Treated Effluent Overflow

Treated effluent from the Newtown Creek WWTP is normally discharged to the East River, but during certain high-flow (wet weather) conditions and during high tide, treated effluent from the Newtown Creek WWTP is discharged via the high-relief outfall (NCB-002) to Whale Creek. The flow splitting is based on hydraulic conditions and does not require operator control. Tidal elevation and total plant inflows determine when the treated effluent is directed toward the overflow (Greeley and Hansen et al. 2010). During high flows (wet weather), the discharge from NCB-002 may include treated combined flows from areas in Brooklyn and Manhattan that do not drain to the Study Area via CSOs (NYCDEP 2007a, 2011a). This discharge is included as a CSO on the Newtown Creek WWTP permit (SPDES Permit No. NY-0287890, effective August 1, 2015).

3.2.8.2.3 Individually Permitted Discharges

Eight individually permitted discharges to Newtown Creek were identified in USEPA online databases and are shown in Figure 3-23; seven of these discharges[55] are currently in use. These discharges are from industrial and municipal facilities and include treated effluent from groundwater remediation and dewatering systems, stormwater, water discharged from secondary containment systems, and hydrostatic test water (USEPA 2012a, 2012b, 2012c). The degree of treatment is variable and documented in each individual SPDES permit:

- **Con Edison – 11th Street Conduit (DAR No. 110)**. The site is authorized under SPDES Permit No. NY0201138 to discharge groundwater and stormwater that has infiltrated into an on-site utility conduit vault. Flows are routed through an oil-water separator prior to discharge.
- **ExxonMobil Greenpoint Remediation Project (DAR No. 53; two discharges)**. ExxonMobil operates two groundwater treatment systems, referred to as the on-site

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[55] The discharge from Getty Terminals (DAR No. 47) regulated under SPDES Permit No. NY0028452 was also regulated in 2012; however, following the sale of the property and change in site use in 2014, this discharge is no longer permitted.

and off-site systems. Filtered stormwater is combined with treated groundwater effluent from the on-site treatment system and discharged to Newtown Creek under individual SPDES Permit No. NY0267724 via Outfall NY0267724-001. Treated groundwater effluent from the off-site system is also discharged to Newtown Creek under SPDES Permit No. NY0267724, via outfall NY0267724-002. Both the on-site and off-site systems involve multiple treatment technologies. The on-site treatment system includes an oil-water separator and sand filtration. The off-site treatment system includes aeration, sand filtration, and catalytic oxidation.

- **Buckeye Pipeline Facility (DAR No. 106)**. Outfall BB-0200441-001 is located on this site's shoreline and is authorized to discharge treated groundwater under SPDES Permit No. NY0200441. Recovered groundwater is pumped through granular activated carbon (GAC) prior to discharge to Newtown Creek.[56]
- **Motiva Brooklyn Terminal (DAR No. 50)**. Stormwater runoff from the site flows into an oil-water separator, prior to discharging to Newtown Creek via Outfall NY0006131-001, which is authorized under SPDES Permit No. NY0006131 to discharge stormwater runoff and hydrostatic test water.
- **KM Phoenix Holdings LLC (former BP Products N America Brooklyn Terminal [DAR No. 48])**. KM Phoenix Holdings LLC acquired the Brooklyn Terminal, and Kinder Morgan Liquids Terminals LLC began operations at the terminal on February 1, 2016. Prior to February 1, 2016, during the period when BP operated this terminal, stormwater was collected and treated with two oil-water separators and two GAC treatment units prior to discharge to Newtown Creek via Outfall NCB-0004596-001, as then authorized under SPDES Permit No. NY0004596. In addition to stormwater, this permit also authorized the discharge of secondary containment water and hydrostatic test water. BP is currently not involved with Kinder Morgan's operation of the terminal.
- **Queens District 5/5a Garage (DAR No. 45)**. Stormwater collects in catch basins and trench drains located throughout the site and is conveyed to an oil-water separator for treatment before discharge to Newtown Creek via Outfall NY0200841-002, under SPDES Permit No. NY0200841.

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[56] Remedial pumping at the Buckeye Pipeline Facility (DAR No. 106) occurred from 1987 to October 2012. In March 2015, NYSDEC discontinued the SPDES permit due to the removal of the treatment system (GES 2017).

- **Bayside Fuel Oil Depot (DAR No. 51)**. Stormwater is conveyed through an oil-water separator and discharged to English Kills via Outfall NCB-0007641-001, which is authorized under SPDES Permit No. NY0007641 to discharge stormwater and hydrostatic test water (KAR Engineering 2012).

3.2.8.2.4 Other Stormwater Discharges

More than 300 private and municipal outfalls have been documented along Newtown Creek and its tributaries, some of which may be abandoned or no longer in use.[57] Of these, 22 are CSO outfalls, 1 outfall is the treated effluent overflow from the Newtown Creek WWTP, and 4 outfalls discharge treated effluent from groundwater remediation and dewatering systems. The rest are stormwater outfalls (see Figures 3-19a through 3-19j and Figure 3-23). Stormwater discharges include point source discharges and overland flow discharges from individual sites directly to the creek, from municipal infrastructure to the creek, or from a combination of both.

Stormwater discharges to the Study Area are from sites that have a multisector general permit (MSGP) and municipal discharges for which there is an NYC MS4 permit,[58] as well as stormwater discharges that are not regulated. MS4 and major stormwater outfalls discharge runoff from multiple sites, and in some cases, large portions of the Newtown Creek drainage basin discharge to the creek at a single outfall location. In several areas within the drainage basin, stormwater runoff from private sites and roadways drains to MS4 infrastructure and is subsequently discharged to the creek. MS4s are owned by NYC and under the jurisdiction of NYSDEC (SPDES Permit No. NY0287890).

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[57] The development of the point sources inventory is discussed in detail in Section 2.1.2 of Appendix E.

[58] Individual permits are specifically developed for the needs of a single facility. General permits are written to apply to multiple discharges within a type of activity or geographic area that have similar environmental impacts. In New York State, an MSGP (No. GP-0-12-001) for stormwater discharges associated with industrial activity has been issued by NYSDEC. The MSGP is applicable to 31 categories of industrial activities, including scrap recycling and waste recycling facilities, land transportation, and warehouse facilities. Facilities that fall within any of these 31 categories may apply for coverage under the permit. On August 1, 2015, NYSDEC issued a new permit (SPDES Permit No. NY0287890), which covers NYC-owned MS4 discharges (NYSDEC 2016b).

3.2.8.3 Regulation of Historical and Current Discharges

Prior to 1938, when the first City-wide unified rules and regulations for the use of the municipal sewer system were adopted, individual boroughs were responsible for regulating discharges to municipal sewers. These early regulations prohibited discharges that could damage the infrastructure or threaten public safety (e.g., high temperatures and extreme flows; Imbelli et al. 1968). In 1963, the first City-wide industrial waste control program was promulgated, requiring permits for the discharge of industrial waste and otherwise toxic substances and imposing a sewer surcharge on industrial waste discharges (Imbelli et al. 1968).

Regulation of private and municipal discharges to the creek was increased with the passage of the CWA in 1972 and subsequent amendments. Following the passage of the 1972 CWA and the 1977 amendments, NYSDEC was responsible for regulating the discharge of pollutants from point sources and non-point sources (e.g., overland flow) into waters of the state. SPDES permits for industrial discharges to the Study Area, like the Shell Oil Company's Marketing Terminal located on Paige Place, were issued in the 1970s (SPDES Permit No. 000613; effective date June 17, 1974; NYSDEC 1974b). Permitted facilities were required to monitor the discharge for parameters identified in the permit and report the monitoring data to regulatory agencies in discharge monitoring reports (DMRs).

In 1988, NYSDEC began issuing SPDES permits to NYCDEP for the City of New York's 14 WWTPs, including the Newtown Creek WWTP (SPDES No. NY0026204) and the Bowery Bay WWTP (SPDES No. NY0026158). These permits established limits for the discharge of effluent from the plants (to the East River and Flushing Bay) and best management practices (BMPs) for CSOs, including the Industrial Pretreatment Program, which regulates industrial wastewater discharges to the municipal sewer. The permits also contained conditions requiring the planning and implementation of strategies to abate CSOs (1988 permit; 1992 CSO Order). NYCDEP was unable to fulfill the requirements of the permits to initiate and complete the required CSO plans, including the plan for Newtown Creek, in compliance with the deadlines established in the permits. In June 1992, NYCDEP entered into a consent order with NYSDEC (NYSDEC 1992). The order was amended on September 19, 1996, to include provisions addressing catch basin maintenance. The 1992 and 1996 orders were superseded on January 14, 2005, by a new consent order issued to address numerous non-conformances to the earlier orders. The 2005 order required implementation of projects and facilities plans culminating in an LTCP for CSOs. The 2005 order was modified on April 14, 2008; September 3, 2009; and March 8, 2012. In 2005, NYC also signed a Memorandum of Understanding with NYSDEC to facilitate water quality standard reviews (NYSDEC 2012b; NYCDEP 2011a).

As part of these agreements, NYCDEP is developing 10 waterbody-specific LTCPs, plus one City-wide LTCP, to reduce CSO discharges and improve water quality in NYC's waterbodies and waterways. The LTCPs are intended to focus on identification of appropriate CSO controls necessary to achieve waterbody-specific water quality standards, consistent with the federal CSO policy and the water quality goals of the CWA. The Newtown Creek LTCP is discussed in Section 3.2.8.3.1.

3.2.8.3.1 Combined Sewer Overflow Long-Term Control Plan

The Newtown Creek LTCP (NYCDEP 2017) was approved on June 27, 2018 (NYSDEC 2018a). The LTCP includes plans to construct two "preferred" CSO controls. Timing of CSO controls is an important consideration for the Newtown Creek RI/FS. The first control will reroute the Dutch Kills CSO from Outfall BB-026 through the expanded Borden Avenue Pumping Station (BAPS) to the Newtown Creek WWTP. The Dutch Kills BAPS CSO rerouting project is scheduled to be completed in 2029 (8 years from the writing of this report) and is predicted to reduce annual CSO volume to Newtown Creek by 110 million gallons per year (MGY) (20% CSO flow reduction[59]). The second control will provide underground tunnel storage for CSOs from the three largest Newtown Creek outfalls, which are located in English Kills (Outfall NCB-015), East Branch (Outfall NCB-083), and Maspeth Creek (Outfall NCQ-077). The storage tunnel will include a tunnel dewatering pumping station that will pump the CSO waters stored in the tunnel to the Newtown Creek WWTP for treatment after wet weather conditions have ended. The storage tunnel project is scheduled to be completed in 2042 (21 years from the writing of this report) and is predicted to reduce annual CSO volume to Newtown Creek by 584 MGY (an additional 61% flow reduction, for a total of 69% CSO volume reduction from current levels).

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[59] CSO flow reductions referenced in this section do not include discharge volume from the Whale Creek treated effluent overflow, which does not appear to be directly addressed by the LTCP.

The uncontrolled CSO discharges that will remain during and after the proposed LTCP action is implemented—along with discharges from the Newtown Creek WWTP treated effluent overflow and stormwater discharges—will contribute to ongoing external inputs to Newtown Creek. NYCDEP estimates that total discharges from point sources (including CSOs) and overland flow discharges into the creek currently amount to a total baseline volume of 3,743 MGY, including 1,162 MGY from CSOs and 1,650 MGY of treated effluent from the Newtown Creek WWTP (discharged at Whale Creek).

As part of the LTCP process, a series of infrastructure projects have already been completed by NYC in the Newtown Creek sewershed to address CWA requirements, including the installation of enhanced aeration, bending weirs, floatables control, and green infrastructure. The status of these elements is captured in the quarterly progress reports submitted to NYSDEC (NYCDEP 2018a).

In an effort to meet water quality standards for DO, NYCDEP has installed an aeration system in English Kills, East Branch, and the upper portion of the Turning Basin (NYCDEP 2018a). The aeration system consists of sections of air header piping that are connected to a series of diffusers that distribute air into the water column. The infrastructure for the diffusers and piping is typically less than 1 to 2 feet tall and rests on the bottom of the creek (USACE 2016). The aeration system only operates during certain months of the year when DO levels are below regulatory requirements of 3 mg/L specified by the CWA. As part of the permit approval process for the system, NYCDEP will be required to remove the infrastructure associated with the aeration system in the creek for any of the following reasons:

- Maintenance dredging required by USACE
- Obstruction of vessel traffic or interference with navigation or adjacent facilities
- USEPA-required remedial activities within the creek (USACE 2013a)

Future expansions of the aeration system were originally planned to also cover Dutch Kills and lower Newtown Creek, but those expansions were eliminated in the final LTCP.

3.2.9 Surface Water Classifications, Fish Advisories, and Public Health Assessment

All waters in New York State have been classified based on best use and the nature of the waterbody, as described in the New York Codes, Rules, and Regulations (CRR) Title 6 Part 701 Classifications – Surface Waters and Groundwaters (6 CRR-NY 701).[60] The water quality standards program is a state program that is conducted with federal oversight by USEPA. Water quality standards and effluent limits have been established for several parameters, toxics, and substances and are used by NYSDEC as the regulatory targets to protect the uses identified in Part 701 (see 6 CRR-NY 703 – Surface Water and Groundwater Quality Standards and Groundwater Effluent Limitations). Newtown Creek is classified by the State of New York as a Class SD saline surface water based on state water quality standards (see 6 CRR-NY 890.6 Table I). As noted by NYSDEC, "the best usage of Class SD waters is fishing, and these waters shall be suitable for fish, shellfish, and wildlife survival. In addition, the water quality shall be suitable for primary and secondary contact recreation, although other factors may limit the use for these purposes. This classification may be given to waters that, because of natural or man-made conditions, cannot meet the requirements for fish propagation" (see 6 CRR-NY 701.14). The nearby East River is classified as a Class I saline surface water, where best usages are secondary contact recreation and fishing (see 6 CRR-NY 890.6 Table I). Class I waters are also suitable for fish propagation and survival (see 6 CRR-NY 890.6, 6 CRR-NY 701.13, and 6 CRR-NY 701.14).

Until recently, Class SD waterbodies in New York State, such as Newtown Creek, were not required to be suitable for primary and secondary recreation and were only required to meet water quality standards for fish, shellfish, and wildlife survival. The only numerical criterion for fish, shellfish, and wildlife survival is DO (3 mg/L). As discussed in the LTCP (NYCDEP 2017), much of Newtown Creek and the tributaries consistently do not meet the NYSDEC Class SD fish survival standard for DO concentration (i.e., never less than 3 mg/L) due to CSO discharges.[61] As a result, Newtown Creek was included on the 2006 CWA Section 305(b) list, with identified use impairment to aquatic life (precluded), recreation

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[60] The effective date for Part 701.14 for Class SD saline waters is November 4, 2015.

[61] Table 6-3 of the 2017 LTCP (approved June 27, 2018 [NYCDEP 2018a]) shows that 60% of baseline biological oxygen demand from the creek is from CSOs, with another 20% coming from the treated WWTP effluent discharged at Whale Creek.

(impaired), and aesthetics (stressed due to floatables). Although Newtown Creek was also included on the 2006 New York State total maximum daily load priority list under CWA Section 303(d) with identified impairment to fish propagation due to low DO concentrations, in 2012 the creek was delisted because it is being addressed under the LTCP.

As a result of recent amendments (i.e., November 4, 2015) by NYSDEC to Part 701.14, Class SD waters shall be suitable for both primary and secondary contact recreation and support federal CWA requirements that all waters meet fishable/swimmable goals (6 CRR-NY 701.14). These modifications will require Newtown Creek to comply with additional water quality requirements for total coliform and fecal coliform standards (see 6 CRR-NY 703.4[a] and [b]). The amendments do not affect the DO standard or the best usage of Newtown Creek, but will require compliance with the regulatory standards for total and fecal coliform. The recent amendments by NYSDEC will require the LTCP to address the revised water quality standards, which include criteria for DO, total coliform,[62] and fecal coliform.[63]

In 2014, NYSDOH released a public health assessment for Newtown Creek. This report was prepared to fulfill the congressional requirement that public health assessments be conducted for National Priorities List sites. The report concluded that swimming and other full body immersion could harm people's health, due to biological hazards attributed to CSOs and the industrial nature of the creek. Sampling conducted by NYSDOH indicated that coliform and enterococci bacteria in the creek increased following CSO events. Coliform counts in samples collected in 2010 from four stations on the creek ranged from 10 to 200,000 per 100 mL, and enterococci ranged from 2 to 20,000 per 100 mL (MPN; NYSDOH 2014).

NYSDOH has issued sportfish health advisories for the East River and connected, unobstructed waterways, including Newtown Creek (NYSDOH 2015a). The NYSDOH contaminants of concern for the East River (and Newtown Creek) are PCBs in fish; Cd, dioxin, and PCBs in crab and lobster tomalley; and Cd and PCBs in blue crab meat (NYSDOH 2015a). Sensitive populations (i.e., women of childbearing age and children) are advised to not eat any sportfish species or crabs from the East River and Newtown Creek (NYSDOH

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[62] The monthly median value and more than 20% of the samples, from a minimum of five examinations, shall not exceed 2,400 and 5,000 per 100 mL, respectively.

[63] The monthly geometric mean, from a minimum of five examinations, shall not exceed 200 per 100 mL.

2015a). Women over 50 and men over 15 years old are advised to not eat channel catfish (*Ictalurus punctatus*), gizzard shad (*Dorosoma cepedianum*), white catfish (*Ameiurus catus*), the hepatopancreas (or tomalley) of crabs and lobsters, or the liquid that crabs have been cooked in. This population is further advised that they may eat up to one meal per month (equal to half of a pound) of Atlantic needlefish (*Strongylura marina*), bluefish (*Pomatomus saltatrix*), rainbow smelt (*Osmerus mordax*), striped bass, white perch (*Morone americana*), carp (*Cyprinus carpio*), and goldfish (*Carassius auratus*); up to four meals a month of other fish species; and up to four meals per month of blue crab meat (six crabs per meal) from the East River and Newtown Creek (NYSDOH 2015a). NYSDOH also recommends preparation and cooking methods to minimize the ingestion of fish lipids, which may contain higher concentrations of PCBs, dioxins, or other chemicals (NYSDOH 2015b).

3.2.10 Groundwater Use

Groundwater near the Study Area was used as a source of potable water in the late 1800s and mid-1900s. As discussed in Section 3.1.1.2, withdrawals for public water supply and industrial use resulted in a dramatic decline in the water table and resulted in saltwater intrusion in some areas (Cartwright 2002; Misut and Monti 1999). Groundwater withdrawal volumes have continually decreased since the middle of the twentieth century, in part due to increased regulation of groundwater withdrawals and uses, as well as the opening of the NYC water tunnels (Cartwright 2002). Current use of groundwater in the upland areas adjacent to the Study Area is limited to non-potable uses, such as irrigation (Plache 2015).

3.2.10.1 Historical Groundwater Use

By 1889, several water supply companies were withdrawing groundwater in Brooklyn and Queens, including the Citizen’s Water Company of Newtown (established 1893), which operated near Maspeth Creek (see Figure 3-24; MANY 1900). Groundwater withdrawals for use as public drinking water reached approximately 33 MGD in Kings County and 65 MGD in Queens County in 1910 (Buxton et al. 1981; Misut and Monti 1999). A temporary decrease in withdrawals occurred in 1917, coinciding with the opening of the first NYC water tunnel that conveyed water from upstate New York to the city. However, between 1918 and 1930, groundwater use for both public and industrial supply continued to increase

(Buxton et al. 1981; Misut and Monti 1999). In the late 1920s, industrial withdrawals in Kings and Queens Counties were estimated to be 75 MGD (Suter 1937).

The increasing withdrawals coincided with the expansion of the municipal sewer system (discussed in Section 3.2.8.1). Instead of recharging the aquifer, groundwater pumped for use in homes and businesses was typically discharged to the sewer system. As recharge decreased and groundwater withdrawals continued, the water table was depressed, resulting in saltwater intrusion into the UGA in several areas within Kings and Queens Counties, including near Newtown Creek.

Between 1903 and 1936, the water table on the Queens and Brooklyn sides of Newtown Creek dropped approximately 50 feet and 35 feet, respectively (Cartwright 2002). The water table north of Newtown Creek remained above sea level in 1936. However, south of Newtown Creek, the water table was approximately 35 feet below sea level in 1936 (Cartwright 2002). Negative (i.e., below sea level) water-table conditions were mapped in 1936, adjacent to the shoreline of the Study Area, from the approximate location of Maspeth Creek to East Branch, English Kills, and along the southern shoreline of Newtown Creek to the East River. The water-table depression south of the Study Area in 1936 was up to 30 feet below sea level within approximately 0.5 mile south of the upstream end of English Kills (Cartwright 2002). As noted later in this section, the negative water-table elevation south of Newtown Creek persisted until sometime between 1961 and 1974. It is interpreted that, throughout the period with negative water-table elevations along the south shoreline to the Study Area, saltwater intruded into the UGA from Newtown Creek.

By the 1930s, concerns about diminishing groundwater supply resulted in additional regulation of groundwater withdrawals. New York State passed water conservation laws requiring permits for new wells and reinjection of withdrawals above 0.1 MGD. Also in the 1930s, refrigerators and air conditioners began to be used widely (reducing the demand for commercial scale ice production), and the second water tunnel to NYC was completed (Cartwright 2002). In 1937, three large private water supply companies remained in Brooklyn and Queens. These included the Woodhaven Water Supply Company and the Jamaica Water Supply Company, located southeast of Newtown Creek near the border with Nassau County; and the Flatbush Waterworks Company located south of the creek in

Brooklyn (Suter 1937). Franchise areas for these companies were several miles from Newtown Creek and are shown in Figure 3-24.

By 1947, groundwater withdrawals for use as public drinking water ceased in Kings County, primarily because of saltwater intrusion. However, the water-table depression south of Newtown Creek dissipated slowly, and the water table remained below sea level until sometime between 1961 and 1974 (Cartwright 2002). Industrial supply withdrawals fell below 10 MGD in Kings County and 3 MGD in Queens County by 1983 (Misut and Monti 1999). The water table near the Study Area returned to approximate current conditions sometime between 1983 and 1997 (Cartwright 2002). Therefore, saline groundwater underneath Newtown Creek under current conditions could be a result of saltwater intrusion followed by flushing as water levels recover to pre-development levels (Cartwright 2002). However, remediation system pumping adjacent to Newtown Creek (see Section 3.2.10.2) could also increase groundwater salinity by pulling saline surface water into the UGA from Newtown Creek.

By the early 1990s, the only public supply pumping in Kings and Queens Counties was 22 MGD from Jamaica Water Supply Company wells, as shown in Figure 3-24. In 1996, when NYCDEP purchased the Jamaica Water Supply Company, well withdrawals had decreased to 14 MGD, and withdrawals from these wells for drinking water distribution ended in 2007 (NYCDEP 2012a).

3.2.10.2 Current Groundwater Use

The results of groundwater sampling and analyses during Phase 2 indicate that groundwater underlying the Study Area is saline and does not meet the New York State requirements for use as a potable or non-potable water source. NYSDEC classifies groundwater and other waters of the state in accordance with best usage in the interest of the public (NYSDEC 2015a). The three groundwater classifications are described as follows:

- **Class GA fresh groundwater**. Class GA fresh groundwater has a chloride concentration equal to or less than 250 mg/L[64] or a total dissolved solids (TDS)

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[64] Chloride is analyzed using USEPA Method 300.0.

concentration equal to or less than 1,000 mg/L. The best usage of Class GA waters is as a source of potable water.

- **Class GSA saline groundwater**. Class GSA saline groundwater has a chloride concentration of more than 250 mg/L or a TDS concentration of more than 1,000 mg/L. The best usages of Class GSA waters are as a source of potable mineral waters, conversion to fresh potable waters, or as raw material for the manufacture of sodium chloride or its derivatives or similar products.
- **Class GSB saline groundwater**. Class GSB saline groundwater has a chloride concentration greater than 1,000 mg/L or a TDS concentration greater than 2,000 mg/L. The best usage of Class GSB waters is as receiving water for disposal of wastes. (Note that for groundwater to be classified as Class GSB, NYSDEC must make a determination that adjacent and tributary groundwater and the best usages thereof will not be impaired by such classification.)

As described in the USEPA-approved BHHRA (Appendix H, Section 2.5.3), by default, Class GSA is assigned to saline groundwater in New York State (6 CRR 701.18). Thus, this is the groundwater classification in the Study Area.

A permit from the New York City Department of Health and Mental Hygiene is required for the installation, drilling, replacement, or operation of a water well, water well pump, or well pumping equipment used to supply water to buildings in NYC. Permits are obtained for potable or non-potable wells and must be renewed annually (NYC [date unknown]). Applicants for potable wells must establish that the municipal water supply is not accessible and that the water quality of the proposed well will meet the standards outlined in Subpart 5-1 of the New York State Sanitary Code (NYCDHMH [date unknown]). In addition, a Long Island Well Permit is also needed from NYSDEC to withdraw water from Long Island aquifers.

A list of permitted water wells was obtained from New York City Department of Law (Plache 2015). A well permit search was also conducted at the Region 2 NYSDEC office in 2012. There are no permitted potable supply wells near Newtown Creek. There are the following permitted, non-potable supply wells (see Figure 3-25):

- Structural Processing Corp., Permit No. 40421099

- Cornell Beverages Inc., Permit No. 40421113
- Mount Olivet Cemetery, Permit No. 40421400
- 7 Islands Car Wash Inc., Permit No. 40423200
- Mount Carmel Cemetery, Permit No. 40421043
- Lebanon Cemetery Association, Permit No. 40421060

Temporary and permanent dewatering wells, including groundwater extraction wells at remediation sites, also exist near the Study Area (see discussion in Section 4.5.2.1 of Appendix F). Groundwater withdrawals from dewatering wells may be discharged directly to the municipal sewer or directly to surface water (e.g., Newtown Creek or East River). In some cases, groundwater must be treated prior to discharge to meet applicable effluent requirements. Groundwater discharges to the municipal sewer must be approved by NYCDEP. Discharges to surface water are regulated by NYSDEC (NYCDHMH [date unknown]).

The Metropolitan Transportation Authority (MTA) extracts groundwater to dewater its subway tunnels and subway stations and is likely the largest industrial withdrawer of groundwater in Kings and Queens Counties (Misut and Monti 1999). The MTA currently pumps between 5.8 and 13 MGD (2,117 and 4,745 MGY, respectively) of groundwater from all of its subway tunnels (MTA 2008, 2009). Based on data received from the MTA in April 2018 (Schmidt 2018a, 2018b; Kwan 2018), the MTA pumps approximately 5.5 MGD from 14 pump rooms and six deep wells near the Study Area. Specific locations and pumping rates for MTA pumping plants are discussed in detail in Appendix F, Section 5.1.2.2. The majority of this pumping, 5.2 MGD, occurs at the Marcy/Crosstown deep wells located southwest of the Study Area, approximately 1.25 miles from the head of English Kills.

3.2.11 Historical Spills

Approximately 90% of the annual 16,000 environmental releases reported across the state through 2012 to NYSDEC involve petroleum products (NYSDEC 2012c). State law requires the spiller or responsible party to notify NYSDEC of petroleum spills within 2 hours of discovery, except for those spills that meet the following criteria:

- Quantity is known to be less than 5 gallons.
- Spill is contained.

- Spill has not and will not reach the State of New York's water or land.
- Spill is cleaned up within 2 hours of discovery (NYSDEC 2012c).

The response by the NYSDEC Spill Response Program, including immediate response as well as continued cleanup, varies depending on the type of material spilled and the damage caused. According to the NYSDEC Spill Incidents Database, spill reporting and cleanup is a relatively recent practice beginning in January 1978. Historical operational practices would have included spills to surface water that, today, would no longer be standard or acceptable practice.

The NYSDEC Spills Incident Database contains information for reported spills to surface water dating back to 1978 and details the cause, location, and source of a spill and subsequent investigations, if any. The information available in the database is variable. It can be difficult to decipher where a release occurred, and the source, cause, or quantity of a spill may be unknown. For example, the first spill recorded in the NYSDEC Spills Incident Database in the Brooklyn Borough for Newtown Creek was reported on April 1, 1978. The material spilled is not identified, the amount spilled is unknown, and the location provided is a street that runs the entire length of Brooklyn. The source and cause of the spill was not included in the reported incident, and the spill record was closed a year later without clear resolution. Remedial actions executed regarding the spill are unknown. In the 39 years since the first spill incident was recorded in Brooklyn in the NYSDEC Spills Incident Database, there have been approximately 70 documented spills designated as "affecting surface waters" in Newtown Creek or its tributaries. Using a dataset of spills from 1983 to 2017 obtained from the NYSDEC FTP site, sites with documented releases to Newtown Creek or its tributaries are included in Table 3-1 (see also Section 4.2.2 of the draft DAR in Appendix J). For the majority of these spills, the sources and/or quantities are undocumented. Two known spills to the Study Area not found in the NYSDEC Spills Incident Database include incidents in 1978 and 2014; these spills are described in the subsequent paragraphs.

In 1978, the U.S. Coast Guard (USCG) first detected signs of an oil release entering Newtown Creek. A subsequent investigation concluded that the area of the release under the Greenpoint area was in excess of 52 acres, and the total release volume, as estimated in 1979, was approximately 17 MG of petroleum product (USEPA 2007a). Petroleum product recovery operations are in place within four distinct areas, including the former ExxonMobil Greenpoint Remediation Project (DAR No. 53), the BP Products N America Brooklyn Terminal (DAR No. 48),[65] the commercial/industrial/residential area southwest of the BP Products N America Brooklyn Terminal known as the "Off-Site Area," and the site of the Empire Merchants/Former Paragon Oil Terminal (DAR No. 200) (USEPA 2007a). ExxonMobil has been actively engaged in remediation activities in Greenpoint on a continuous basis since 1979. These activities were initiated voluntarily by ExxonMobil, which, after the 1978 discovery by USCG, began product recovery operations on its own property and participated in the Meeker Avenue Task Force recovery operations at the foot of Meeker Avenue. Petroleum product recovery has exceeded 13 MG of product (as of January 2019), with the majority of product recovery from systems installed and operated by ExxonMobil. The hydraulic control provided by ExxonMobil's systems has the beneficial effect of addressing multiple other sources of off-site contamination that are not attributable to ExxonMobil, but are present in the Study Area due to the long industrial history of the area.

In the spring of 2014, petroleum sheens were seen on Newtown Creek—specifically the northwest section of the creek running from the Pulaski Bridge to Whale Creek and up into Dutch Kills. In September 2014, NYSDEC received an anonymous tip that oil was being dumped into a drain behind a business near the Borden Avenue Bridge (NYT 2014). NYSDEC, USGS, and a private environmental contractor deployed absorbent and containment booms both upstream and downstream of the source. Forensic evidence was also collected for later prosecution of the alleged perpetrator (Brownstoner 2014).

Other remedial efforts in the Newtown Creek groundwater recharge area that have been installed to address contamination potentially resulting from historical releases include groundwater extraction and LNAPL recovery systems at the Morgan Oil (DAR No. 60), Motiva Brooklyn Terminal (DAR No. 50), and Malu Properties/Former Ditmas Oil/Former Gulf Oil (DAR No. 123) sites along the southern shore of the Study Area; and at the Former Laurel Hill Site (DAR No. 16), Waste Management of NY/Steel Equities (formerly POW) (DAR No. 56), Quanta/RAD II (DAR No. 39), RAD I (DAR No. 41), and Buckeye Pipeline

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[65] BP Products N America Brooklyn Terminal was sold to Kinder Morgan Operating L.P. "D," a Delaware limited partnership effective February 2, 2016. Nonetheless, to maintain consistency with the draft DAR, this facility is referred to as "BP Products N America Brooklyn Terminal."

Facility (DAR No. 106) along the northern shore of the Study Area. Each of these sites and systems is discussed in more detail in the following, although it is noted that this list is not exhaustive (see the draft DAR for a comprehensive evaluation of historical spills and remediation actions):

- **Morgan Oil (DAR No. 60) (see Table 5-3, Figure 5-1, Section 7.10, and Appendix B [specifically, the relevant upland site summary] of the draft DAR, as well as the updated upland site summary, in Appendix J of this RI Report)**. A number of spills to English Kills have been associated with the site, with the most recent occurring in November 2017 (Spill No. 1707515). In November 1992, an unknown amount of fuel oil was spilled to groundwater, and the USCG observed oil near the site in English Kills (NYSDEC Spill No. 9209135). Due to the spill, an RI was conducted in 1992/1993, with additional soil and groundwater investigations conducted in the late 1990s and early 2000s. Morgan Oil installed and operated a groundwater extraction system in June 1995 and continued until operations were ceased in February 2004. The groundwater extraction system at this site used 10 extraction wells outfitted with ejector pumps and a GAC system used to treat effluent. In 1997, oil was observed migrating through the bulkhead into English Kills (EDR 2010). Soil removal actions under an Order on Consent with NYSDEC were conducted in 2012 and 2013, including several AST removals. As of 2016, NYSDEC required weekly groundwater monitoring, inspection and maintenance of the absorbent boom in Newtown Creek, and installation of additional monitoring wells adjacent to the creek (McGaha Consulting 2016).
- **Motiva Brooklyn Terminal (DAR No. 50) (see Table 5-3, Figure 5-1, Section 7.1, and Appendix B [specifically, the relevant upland site summary] of the draft DAR in Appendix J of this RI Report)**. NAPL recovery at the site was conducted from 1999 to 2005, using a combined recovery well and cutoff trench recovery system with an automated, pneumatic product recovery system to clean up and remove discharged petroleum from the vicinity of the facility’s vapor recovery unit and AST secondary containment walls. NAPL was recovered using a floating skimmer head and was pumped to an existing 275-gallon storage tank. Four product interceptor trenches were installed near the southern AST dike walls (Handex 2000). Each trench was excavated to approximately 6 feet below grade. Four monitoring wells were also

retrofitted as dike recovery wells. NAPL recovery via pneumatic pumping ceased in 2005, but manual bailing of wells has been ongoing since.

- **Malu Properties/Former Ditmas Oil/Former Gulf Oil (DAR No. 123) (see Table 5-3, Figure 5-1, and Appendix C [specifically, the relevant upland site summary] of the draft DAR in Appendix J of this RI Report)**. There are three documented spills from the site. One was a January 1975 gasoline spill to a secondary containment dike that resulted in approximately 1,000 gallons of gasoline migrating under the dike wall to English Kills (NYT 1975). The spill was immediately contained with booms and removed from the water surface using vacuum trucks. The second was a surface spill by Ditmas Oil into secondary containment in 1990 that reportedly reached English Kills (Spill No. 9006603; closed in February 2017), and the third was a spill from an underground pipeline in 1999 (Spill No. 9812647; closed in September 2003). Characterization and remediation efforts by Ditmas Oil for its releases from historical petroleum storage operations have included decommissioning of approximately 20 tanks and underground piping, the installation of monitoring wells, interim LNAPL recovery, and a bioremediation system, which was installed by current owner Malu Properties, Inc., in June 2005.
- **Former Laurel Hill Site (DAR No. 16) (see Table 5-3, Figure 5-1, Section 5.6.1, and Appendix C [specifically, the relevant Respondent site summary] of the draft DAR in Appendix J of this RI Report)**. In 2005, a groundwater collection, containment, and treatment system was installed at the Former Laurel Hill Site (DAR No. 16) to address groundwater contamination resulting from historical operations at the site, which included chemical production, copper smelting, and copper refinery operations between the 1870s and 1984. The system consists of an interlocking steel sheetpile barrier wall with eight groundwater collection wells extending the entire length of the property adjacent to Newtown and Maspeth creeks to intercept the shallow groundwater before it reaches the Study Area. Prior to the installation of the groundwater collection system, several other remedial activities took place, including the removal of approximately 12,000 cubic yards of soil and concrete in 1987 and the removal of approximately 21,000 tons of PCB and petroleum-contaminated soils in 2004.
- **Waste Management of NY/Steel Equities (formerly POW) (DAR No. 56) (see Table 5-3, Figure 5-1, Section 5.6.7, and Appendix C [specifically, the relevant Respondent site summary] of the draft DAR in Appendix J of this RI Report)**. At the

site, NAPL was recovered during Interim Remedial Measure (IRM) events initiated by ExxonMobil, beginning in December 2009. The source of the NAPL is unknown, but the viscosity and density of the LNAPL have been documented as being similar to a No. 4 fuel oil. In 2013, ExxonMobil activated an IRM LNAPL recovery system at the site. Additional remedial activities to address NAPL are anticipated, subject to regulatory approval. Additionally, other environmental conditions, including low pH, elevated toluene, and elevated CVOCs, are being addressed by Waste Management under separate Consent Orders.

- **Quanta/RAD II (DAR No. 39) and RAD I (DAR No. 41) (see Table 5-3, Figure 5-1, Sections 7.7 and 7.6, respectively, and Appendix B [specifically, the relevant upland site summaries] of the draft DAR in Appendix J of this RI Report)**. On October 5, 2007, a spill was reported at the site (Lots 279 and 280; NYSDEC Spill No. 0707419). An unknown petroleum product was released and impacted groundwater. The spill is not closed, and the site is in the Brownfield Cleanup Program (NYSDEC 2018a). Permanent NAPL recovery systems with and without thermal enhancement were installed at the Quanta/RAD II and RAD I sites in 2008 (Mitchell and Stetkar 2008); however, no additional information regarding this system was identified in the reviewed documents.
- **Buckeye Pipeline Facility (DAR No. 106) (see Table 5-3, Figure 5-1, and Appendix B [specifically, the relevant upland site summary] of the draft DAR in Appendix J of this RI Report)**. On December 18, 1986, an equipment failure resulted in a gasoline release to surface water (NYSDEC Spill No. 8605941), and Newtown Creek was impacted (EDR 2010). On March 10, 1987, the USCG notified site representatives that ongoing oil seepage through the wooden bulkhead into Newtown Creek had been observed and that the site was in violation of the Federal Water Pollution Control Act. The resulting investigation and remedial activities resulted in the installation of a groundwater/NAPL recovery and treatment system in 1987. The system was designed to recover and remove NAPL gasoline from groundwater by a single 24-inch-diameter recovery well. Recovered NAPL is pumped to a 500-gallon tank, and groundwater is pumped through GAC canisters prior to discharge into Newtown Creek (NEPCCO 1987; EMS 2004, 2009). Ongoing NAPL measurements and recovery are performed and reported quarterly to NYSDEC (EMS 2009). On February 16, 1999, a leak through a flange at a pig exit section of plumbing resulted in

a release of approximately 20 to 30 barrels of unleaded gasoline on the east side of the Buckeye Pipeline Facility (NYSDEC Spill Nos. 9813881 and 9813884). Buckeye Pipeline Facility deployed a 150-foot hard boom and two sorbent booms to contain the LNAPL. The leak was contained and reportedly did not enter the creek directly; however, groundwater was impacted.

4 NATURE AND EXTENT OF CONTAMINATION

4.1 Introduction

This section describes the nature and extent of contamination in the Study Area and reference areas, based on the concentrations of CERCLA hazardous substances in surface sediment, subsurface sediment, native material, surface water, porewater, groundwater, tissue, and air.[66] The discussion is organized by medium and focuses primarily on spatial patterns within the Study Area and comparisons with data collected from the reference areas. Spatial patterns within the Study Area are evaluated by comparing data among the following three reaches: the lower main stem (CM 0 – 2), the upper main stem (CM 2+), and the tributaries (Dutch Kills, Whale Creek, Maspeth Creek, English Kills, and East Branch). Many evaluations are presented for smaller reaches as well (e.g., CM 0 – 1, CM 1 – 2, CM 2+, and each individual tributary). Discussions are not centered around proximity to upland sites or other potential sources, because to do so would be speculative. Chemicals in the Study Area cannot necessarily be attributed to proximate upland site(s) or source(s), including point sources, because of the complex patterns of development and use of upland sites, the complex history of sediment deposition and transport in the Study Area, dredging and modifications to the course of the creek over time, and a lack of upland data for many sites.

4.1.1 Data Used in the Evaluations

The project data are described in Section 2. The data used in this RI Report are a compilation of several different datasets, including data from Phase 1 sampling, data from Phase 2 sampling, data from Part 1 of the FS,[67] and data collected by others that have been approved by USEPA for use in the RI.

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[66] Contamination refers more generally to CERCLA hazardous substances that are the focus of this RI Report, as well as other chemical and biological constituents that are relevant to this investigation. Other chemical and biological constituents, besides CERCLA hazardous substances, include sulfide, ammonia, and complex hydrocarbon mixtures (which are relevant to the understanding of the biological community of the Study Area; see Section 8.5.3.4.2 of Appendix I).

[67] As discussed in Section 2.1, data from a subset of the FS field program (referred to as Part 1 of the FS or FS Part 1) are included as part of the RI dataset. As such, any general references in this report to the RI dataset implicitly include FS Part 1 data, unless otherwise specified.

This section includes discussion of the reference area data as well. Surface sediment samples were collected during Phase 1 and Phase 2 from the 14 Phase 1 reference areas shown in Figure 2-2. Surface water, porewater, and tissue data were collected during Phase 2 from the four Phase 2 reference areas, also indicated in Figure 2-2. Reference area data are included in figures, along with Study Area data, to facilitate comparison of chemical concentrations. Additional detail concerning the collection of reference area data is provided in Section 2.1.2.

4.1.2 Selection of Contaminants for In-Depth Evaluation

The selection of contaminants for in-depth evaluation in this section of the RI Report is based on the outcome of two comprehensive risk assessments: one that evaluated potential risk to human health (the BHHRA) and one that evaluated potential risk to ecological receptors (the BERA). These risk assessments are presented in Appendices H and I, respectively, and summarized in Section 7 of this report. The eight contaminants (or groups of contaminants) selected for evaluation of nature and extent of contamination were those identified to contribute to risk to varying degrees in the BHHRA or BERA; they are as follows:

- TPAH (17) (see Sections 4.1.3.2 and 4.2.5.1 for a definition and further discussion) was selected because elevated bulk sediment and porewater concentrations of PAHs are correlated with observed toxicity to benthic organisms in the Study Area and reference areas.
- TPAH (34) (see Sections 4.1.3.2 and 4.2.5.1 for a definition and further discussion) was selected for the same reasons as TPAH (17).
- C19-C36, representative of hydrocarbons in this carbon range, was selected because elevated bulk sediment concentrations of this class of hydrocarbons are correlated with observed toxicity to benthic organisms in the Study Area and reference areas (Anchor QEA 2021). Other classes of hydrocarbons (i.e., TPH and DRO) exhibited similar correlation with toxicity; the similarity of these classes to C19-C36 and TPAH [34] is evaluated in Section 4.12.1.1).
- TPCB (see Sections 4.1.3.1 and 4.2.5.2 for a definition and further discussion) was selected because TPCB exceeded acceptable human health risk criteria in fish and crab tissue and exceeded acceptable ecological risk criteria to resident benthic fish from porewater exposure and to avian wildlife from dietary exposure.

- 2,3,7,8-TCDD was selected because total dioxins/furans toxic equivalence quotient (TEQ), to which 2,3,7,8-TCDD is a major contributor, exceeded acceptable human health risk criteria in crab tissue and because 2,3,7,8-TCDD exceeded acceptable ecological risk criteria for striped bass.
- Cu was selected because Cu exceeded acceptable ecological risk criteria for blue crab, mummichog, and spotted sandpiper and because its surface sediment concentrations were elevated relative to reference areas.
- Pb was selected because Pb exceeded acceptable risk criteria for the spotted sandpiper.
- Dieldrin was selected because, although it was not identified as a COPC or COPEC, dieldrin tissue concentrations in polychaetes in English Kills exceeded thresholds provided by USEPA for benthic macroinvertebrates.

An overview of the chemical analytes included in the comprehensive RI dataset is provided in Section 4.1.2.1, followed by a summary of the risk assessments in Section 4.1.2.2, which led to the selection of the aforementioned eight contaminants or groups of contaminants (as summarized in Section 4.1.2.3).

4.1.2.1 RI Chemistry Dataset Overview

The RI dataset includes approximately 500 chemical analytes that are being considered in the CERCLA investigation (see Section 2 and Appendix B). These data were collected during Phase 1 and Phase 2 of the RI program and Part 1 of the FS, as well as by other parties (see Section 2, Table 2-1, and Section 4.1.1). The chemical analyses by sampling media are summarized in the following paragraphs. These summaries provide a broad overview of the frequency of detection (FoD) for the classes of chemical analytes, with details provided in the DSRs included in Appendix B (specific tables are referenced for each media). In general, the FoD was higher for metals, PAHs, PCBs, hydrocarbon totals (including TPH, C19-C36, and DRO), and dioxin/furans than other classes of chemicals. Herbicides, VOCs, and SVOCs other than PAHs were infrequently detected, and the FoD for pesticides varied widely.

Sediment: Chemical analyses for surface sediment, subsurface sediment, native material, and sediment trap samples by field program are summarized in Table 2-2b. The FoD for the classes of chemical analytes measured in surface sediment, subsurface sediment, sediment trap, and native material samples is presented in Tables 4-1a to 4-1c, 4-2a and 4-2b, 4-3, and 4-4a and 4-4b, respectively, and is summarized as follows:

- Surface sediment, subsurface sediment, and sediment traps
    - With few exceptions, the FoD was 100% for the entire suite of metals analyzed in surface sediment, subsurface high-resolution cores (segmented every 2 cm), and sediment trap samples.
    - Almost all the individual parent and alkylated PAH compounds were detected with an FoD of 95% and higher (three-quarters with an FoD of 100%) in surface sediment, subsurface high-resolution cores, and sediment trap samples.
    - Approximately three quarters of the individual PCB congeners were detected with an FoD of 100% in surface sediment, subsurface high-resolution subsurface cores, and sediment trap samples.
    - Over half of the individual dioxin/furan congeners were detected with an FoD of 100% in surface sediment, high-resolution subsurface cores, and sediment trap samples
    - VOCs and herbicides were not frequently detected in the Phase 1 field samples. FoD was between 0% and 5% for more than three-quarters of the VOCs and almost all the herbicides. As a result, VOCs and herbicides were not analyzed in Phase 2 sampling.
    - SVOCs other than PAHs were also not detected frequently; the FoD was between 0% and 5% for more than half of the targeted SVOCs in surface sediment, sediment trap samples, and the 1- to 3-foot segmented subsurface cores (SVOCs were not analyzed in the subsurface high-resolution cores). For the remainder of the SVOCs, the FoD ranged from approximately 25% to 99%.
    - Hydrocarbon totals (including TPH, C19-C36, and DRO) were generally detected with an FoD of 90% or higher.
    - FoD for pesticides varied widely from 0% to 100%; in general, the FoD was between 55% and 100% for DDx, chlordanes, dieldrin, hexachlorobenzene, and mirex in surface sediment and sediment trap samples (pesticides were not analyzed in the high-resolution subsurface cores).
- Native material
    - Three-quarters of the metals were detected with an FoD of 100%; for the remaining metals, the FoD ranged from 5% to 99%.

  - FoD for the individual parent and alkylated PAHs ranged from 35% to 100%. FoD was between 80% and 100% for more than half of the individual parent PAHs and more than three-quarters of the alkylated PAHs.
  - FoD for individual PCB and dioxins/furan congeners was lower compared to other sediment sampling media. FoD was less than 50% for more than three-quarters of the dioxin/furan congeners and almost all the PCB congeners.
  - VOCs and SVOCs other than PAHs were infrequently detected. FoD was between 0% and 5% for over three-quarters of both chemical classes. For the remainder of the VOCs, FoD ranged from 6% to 50% and for the SVOCs other than PAHs, FoD ranged from 9% to 89%.
  - Hydrocarbon totals TPH and DRO were detected with an FoD of 91%. C19-36 was infrequently detected, with an FoD of 21%.
  - FoD for pesticides was variable but less than 50% and was lower compared to other sediment sampling media.

Water: Chemical analyses for surface water, porewater, and groundwater samples by field program are summarized in Table 2-2c. The FoD for the classes of chemical analytes measured in surface water, porewater, and groundwater is presented in Tables 4-5a and 5-5b, 4-6, and 4-7, respectively, and is summarized as follows:

- Surface water
  - FoD for metals ranged from 0% for metals such as beryllium (Be) and Ag to 100% for metals such as potassium (K), calcium (Ca), and magnesium (Mg), with the FoD for metals such as Cu, Pb, and Zn ranging from less than 5% up to approximately 70%.
  - FoD for individual parent PAHs and alkylated PAHs varied over a wide range from 0% to 100%, with an FoD of 50% or less for approximately half of the individual parent and alkylated PAHs.
  - FoD ranged from 80% to 100% for just over one-third of the PCB congeners, but was less than 50% for just under one-third of the congeners.
  - FoD was between 0% and 5% for approximately three-quarters of the dioxin/furan congeners; for the remainder, the FoD ranged from 6% to 96%, depending on the congener.
  - FoD for hydrocarbon totals TPH and DRO ranged from 77% to 99% (C19-C36 was not measured in surface water samples).

  - With a few exceptions, the FoD for VOC, SVOCs other than PAHs, herbicides, and pesticides was either 0% or less than 5%.
- Porewater (shallow and mid-depth)
  - FoD for metals in porewater varied, ranging from 0% for metals such as Be to 100% for metals such as K, and Mg, with the FoD for metals such as Cu, Pb, and Zn ranging from less than 20% to approximately 70%.
  - FoD was between 70% and 100% for three-quarters of the individual parent PAHs and alkylated PAHs, and three-quarters of the PCB congeners.
  - FoD was 0% for more than half of the VOCs and SVOCs other than PAHs, with a few exceptions such as phthalates, for which the FoD ranged from 3% to 80%.
  - FoD for C19-C36 (analyzed in mid-depth porewater samples only) was 31% (TPH and DRO were not measured in porewater samples). FoD for pesticides was variable but less than 50% for more than three-quarters of individual pesticides.
  - Porewater samples were not analyzed for dioxins/furans or herbicides.
- Groundwater
  - FoD for metals in groundwater varied, ranging from 0% for metals such as Be and Ag to 100% for metals such as K, Ca, and Mg, with the FoD for metals such as Cu, Pb, and Zn ranging from 30% to 91%.
  - FoD was between 80% and 100% for almost all the individual parent PAHs and alkylated PAHs.
  - FoD for PCB congeners ranged from 0% to 100%, with the FoD for more than half ranging from 80% to 100%.
  - FoD was either 0% or less than 5% for more than half of the VOCs and over three-quarters of the SVOCs other than PAHs.
  - FoD for C19-C36 was 22% (TPH and DRO were not measured in groundwater samples).
  - FoD for pesticides varied, ranging from 2% to 57%.
  - Groundwater samples were not analyzed for dioxins/furans or herbicides.

Tissue: Chemical analyses for tissue by field program are summarized in Table 2-2d. The FoD for the classes of chemical analytes measured in tissue is presented in Table 4-8 and is summarized as follows:

- FoD for metals, including methylmercury, was at or close to 100% for all fish, crab, bivalves, and polychaetes. The one exception was silver, for which the FoD ranged from 0% to 100%.
- FoD for individual PAHs ranged from 0% to 100% in mummichog and bivalves, 8% to 100% in blue crab, 14% to 100% in striped bass, 42% to 100% in Atlantic menhaden, and 62% to 100% in polychaetes (tissue samples were not analyzed for alkylated PAHs).
- FoD was between 80% and 100% for more than three-quarters of the PCB congeners in fish, crab, and polychaetes. For bivalves, less than half of the PCB congeners were detected with an FoD between 80% and 100%.
- FoD for the dioxin/furan congeners was between 20% and 100% for most biota; for blue crab, more dioxin/furan congeners were detected at an FoD of 100% than for any other biota tissue samples.
- FoD was 0% for more than half of the pesticides analyzed for in biota samples, with the exception of DDx, chlordanes, and dieldrin in fish and crab, for which the FoD was at (or close to) 100%.
- Tissue samples were not analyzed for petroleum hydrocarbons, other than PAHs.

Air: Chemical analyses for the air sampling program are summarized in Table 2-2e. The FoD for the classes of chemical analytes measured in air is presented in Table 4-9 and is summarized as follows:

- FoD was zero for eight of the nine PCB Aroclors analyzed, with an FoD of 8% for the remaining Aroclor.
- FoD for VOCs ranged from zero for VOCs such as 1,1-dichlorethane, 1,2,4-trichlorbenzene, and benzyl chloride to 100% for VOCs such as freons, benzene, and methylene chloride.
- Air samples were not analyzed for any other contaminants.

The information on FoD was used, along with detected concentrations, in the risk assessments to identify contaminants requiring further evaluation, as described in Section 4.1.2.2.

4.1.2.2 Risk Assessment Overview

The analytical data collected for surface water, surface sediment, porewater, tissue, and air were used in the risk assessments to evaluate whether there are risks to human health and the environment (aquatic life and wildlife) from exposure to site-related releases of hazardous substances in the various media. This section provides an overview of the risk screening and baseline risk analyses performed in the BHHRA and BERA that resulted in selection of the eight contaminants (or groups of contaminants) for more detailed nature and extent evaluation. A more complete summary of the BHHRA and BERA is presented in Sections 7.1 and 7.2, respectively, with details provided in the USEPA-approved risk assessment reports (see Appendix H for the BHHRA and Appendix I for the BERA).

The BHHRA evaluated potential risks to humans who may be exposed to site-related contaminants through various activities (e.g., recreational and occupational use). The BERA evaluated potential risks to aquatic life and wildlife with different site use characteristics, different feeding strategies, and at different ecological trophic levels.

4.1.2.2.1 Risk Screening

Both risk assessments included a screening step to potentially reduce the number of contaminants to be evaluated in detail in the subsequent baseline analyses. The purpose of the risk screening step was to use conservative exposure assumptions and screening levels (SLs) (i.e., toxicity thresholds) to ensure that no contaminants get erroneously eliminated from further evaluation. Different hierarchies of SLs were used depending on the availability of SLs for contaminants. Details of the human health risk screening are presented in Section 3.3 of Appendix H, and details of the ecological risk screening are presented in Section 5 of Appendix I.

For both risk assessments, the risk screening consisted of the following steps:

- Eliminate essential nutrients (e.g., sodium [Na], K, and Fe), conventional chemicals (e.g., nitrates and nitrites), and chemicals used for forensic analysis (e.g., isoprenoids and thiophenes).
- Eliminate contaminants with an FoD less than 5% from further evaluation, unless the reporting limits (RLs) are greater than the SLs, in which case the contaminants were addressed in an uncertainty analysis.

- For the remaining contaminants, retain those with maximum detected concentrations exceeding SLs, i.e., hazard quotients (HQs) greater than 1 for further evaluation.
- For the human health risk screening, retain detected contaminants that are known human carcinogens.
- For the ecological risk screening, retain contaminants with HQs greater than 1 based on maximum concentrations, and then re-screen and retain contaminants if HQs are greater than 1 based on the 95% upper confidence limit of the mean (95% UCL) concentration for sediment, water, and tissue, or total daily intake (TDI) through the diet.

COPCs identified in human health risk screening are presented in Section 7.1.1 and Table 7-1, while COPECs identified in the ecological risk screening are presented in Section 7.2.3 and Table 7-12.

As detailed in Table 7-1, for human health, COPCs included 44 chemicals for surface sediment, 23 for surface water, 9 for ambient air, 23 for blue crab tissue, 15 for striped bass tissue, and 12 for white perch tissue. For surface sediment and surface water, these consisted of metals, cyanide, individual PAHs, pesticides, PCBs, dioxins/furans,[68] and a limited number of VOCs and SVOCs; for tissue these consisted of metals, individual PAHs, pesticides, PCBs, and dioxins/furans; and for air, these consisted of VOCs and one PCB Aroclor.

As detailed in Table 7-12, for all ecological receptors, COPECS included 32 chemicals for surface sediment, 5 for surface water, and 13 for tissue when using the USEPA Region 2 critical body residues (CBRs). No COPECs were identified based on the NCG CBRs. The number of COPECs varied by media and exposure pathway, depending on the specific feeding guild and trophic level being evaluated. For surface sediment, these consisted of metals, PAHs, pesticides, PCBs, and a limited number of VOCs and SVOCs; for surface water, these consisted of metals and cyanide; and for tissue, these consisted of metals, high-molecular-weight polycyclic aromatic hydrocarbon (HPAH), TPAH (17), pesticides, PCBs, and dioxins/furans.

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[68] For both the human health and ecological risk screening, PCBs and dioxin/furan counts do not include the individual congeners.

4.1.2.2.2 Baseline Risk Analysis

Following the risk screening, both risk assessments evaluated the COPC/COPECs, identified as a result of the risk screening step, in greater detail, with respect to exposure and effects, as applicable. Details of the human health baseline analyses are presented in Sections 4 through 8 of Appendix H, and details of the ecological risk baseline analyses are presented in Sections 6 through 14 of Appendix I. Summaries of the risk assessments are also provided in Sections 7.1.2 through 7.1.4 of this report for human health and Sections 7.2.2 through 7.2.6 of this report for ecological receptors.

Human Health

The BHHRA concluded that the only unacceptable human health cancer risks or noncancer hazards were associated with consumption of striped bass, white perch, and blue crab by recreational anglers and crabbers, as well as noncancer hazards for exposure to surface sediments as a result of general construction work along the shoreline of the Study Area (e.g., bulkhead repair) (see Table 7-2). The COPCs contributing to these risks are as follows (see Tables 7-3 to 7-6):

- For the recreational consumption of striped bass and white perch fillet, the primary contributors to both cancer risks and noncancer hazards were total nondioxin-like PCB congeners and TPCB congener TEQ.
- For the recreational consumption of blue crab muscle and hepatopancreas, the primary contributors to both cancer risks and noncancer hazards were total nondioxin-like PCB congeners, TPCB congener TEQ, and total dioxin/furan TEQ.
- For the general construction worker, the primary contributor to noncancer hazard is total nondioxin-like PCB congeners.

Ecological Receptors

As a result of the baseline ecological analyses, the following COPECs were identified by receptor/receptor category (see Table 7-13):

- Aquatic life – No COPECs were identified, except for cyanide for which the Study Area-wide HQ was 1.1 due to two samples (out of the 360-plus analyzed) that were considered outliers (one in Dutch Kills and one in English Kills). If these two data points are not included in the dataset, the Study Area-wide HQ is 0.8.

- Benthic macroinvertebrates –
  - TPAH (34) was identified as a COPEC based on porewater chemistry and also based on the correlation between TPAH (34) bulk sediment concentrations and observed toxicity.
  - Although not initially identified as a sediment COPEC as a result of the screening level risk assessment, further evaluation by USEPA after the BERA was finalized identified certain classes of hydrocarbons, including C19-C36, TPH, and DRO, in addition to TPAH (34), in bulk sediment as potentially contributing to observed toxicity in the 28-day laboratory toxicity tests. Based on this additional evaluation, USEPA concluded that TPAH (34) and C19-C36 in bulk sediment best represent the observed toxicity to benthic invertebrates (as summarized in Anchor QEA 2021), and TPAH (34) and C19-C36 are included in the discussions of nature and extent in this section. Comparisons between TPAH (34) and C19-C36 with other hydrocarbon sums (i.e., TPAH [17], TPH, and DRO) are provided in Section 4.12.1.1.
  - Porewater concentrations for Cu, Pb, and Zn also exceeded effect thresholds; however, other lines of evidence indicate that metals are generally not bioavailable in sediment porewater.
  - TPAH (17) and TPCB were identified as COPECs based on tissue chemistry and the use of USEPA Region 2 CBRs.
- Bivalves – TPAH (17) and TPCB were identified as COPECs based on tissue chemistry and the use of USEPA Region 2 CBRs.
- Blue crab – TPCB and Cu were identified as COPECs based on tissue chemistry and the use of USEPA Region 2 CBRs.
- Fish – TPCB, Cu, 2,3,7,8-TCDD, and total dioxin/furan TEQ (fish) were identified as COPECs based on tissue chemistry and the use of USEPA Region 2 CBRs, and/or based on a dietary pathway. TPAH (34) and Cu, Pb, and Zn were identified as COPECs based on porewater chemistry for benthic fish, although other lines of evidence indicate that metals are generally not bioavailable in sediment porewater.
- Mummichog – TPCB congeners and Cu were identified as COPECs based on tissue chemistry and the use of USEPA Region 2 CBRs. Cu was identified as a COPEC based on a dietary pathway evaluation. TPAH (34), TPCB congeners, Cu, Pb, and Zn were identified as COPECs based on porewater chemistry for a benthic fish inhabiting the Turning Basin, Maspeth Creek, and English Kills. While not directly applicable to

benthic fish, measurements of SEM and AVS in bulk sediment indicated low bioavailability for these metals with Σ SEM – AVS results less than zero. Sequential extraction also supported the lack of metals bioavailability.

- Avian wildlife – TPCB, Cu, and Pb were identified as COPECs based on a dietary pathway evaluation.
- Green heron, black-crowned night heron, and belted kingfisher – TPCB congeners were identified as a COPEC based on a dietary pathway evaluation.
- Double-crested cormorant and raccoon – No COPECs were identified.

Although dieldrin was not identified as a COPEC, spatial examination of the tissue data in the BERA noted that the concentrations of dieldrin in polychaetes, bivalves, blue crab, striped bass, and mummichog increased toward the upper reaches of the Study Area.

4.1.2.3 Selected Contaminants

As a result of the baseline risk analyses, TPAH (17), TPAH (34), C19-C36, TPCB,[69] 2,3,7,8-TCDD,[70] Cu, and Pb were selected to characterize the nature and extent of environmental impacts in the Study Area. As noted previously, although dieldrin was not identified as a COPC or COPEC in the baseline risk assessments, it is included in the nature and extent evaluation on a more limited basis, with information presented for surface sediment and tissue.

These eight contaminants (or groups of contaminants) were determined to be COPCs/COPECs or were identified to contribute to risk or toxicity (to varying degrees), based on the risk analyses described in Sections 4.1.2.2.1 and 4.1.2.2.2 and summarized as follows:

- TPAH (17) exceeds acceptable risk criteria in bivalve and benthic macroinvertebrate bivalve tissue (see Sections 7.3 and 8.2.3 of Appendix I, respectively).
- Elevated TPAH (34) concentrations in porewater and bulk sediment are correlated with toxicity to benthic organisms in some portions of the Study Area, as discussed in the BERA (see Section 8.3.3.3 of Appendix I).

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[69] TPCB is selected to represent nondioxin-like and dioxin-like PCB congeners evaluated in the BHHRA, and TPCB congeners evaluated in the BERA.

[70] Given that 2,3,7,8-TCDD is a major contributor to total dioxin/furan TEQs, 2,3,7,8-TCDD is the congener selected for the nature and extent characterization.

- Elevated C19-C36 concentrations in bulk sediment are correlated with toxicity to benthic organisms throughout the Study Area, based on additional evaluations completed by USEPA subsequent to the finalization of the BERA.
- TPCB exceeds acceptable risk criteria, as discussed in the BHHRA and BERA (see Section 6 of Appendix H and Sections 14.2 and 14.3 of Appendix I, respectively), as follows:
  - TPCB in fish and crab tissue results in risks that are greater than the USEPA acceptable risk range for cancer risks and exceed noncancer hazard thresholds associated with human consumption of fish and shellfish from the Study Area.
  - Exposure to TPCB concentrations in porewater results in an HQ greater than one for resident benthic fish in some portions of the Study Area.
  - Exposure to TPCB in surface sediment and prey results in an HQ greater than one from dietary exposure for avian wildlife in one portion of the Study Area.
- 2,3,7,8-TCDD and total dioxin/furan TEQs exceed acceptable risk criteria, as discussed in the BHHRA and BERA (see Section 6.2.3.2.3 of Appendix H and Section 10.3.4.1 of Appendix I, respectively), as follows:
  - Total dioxin/furan TEQ in crab tissue results in risks that are greater than the USEPA acceptable risk range for cancer risks and exceed noncancer hazard thresholds associated with human consumption of fish and shellfish from the Study Area. Given that 2,3,7,8-TCDD is a major contributor to total dioxin/furan TEQs in crab tissue, 2,3,7,8-TCDD is the dioxin/furan congener selected for the nature and extent characterization.
  - Exposure to 2,3,7,8-TCDD results in an HQ greater than one from dietary exposure for striped bass in the Study Area.
- Cu is included, because surface sediment concentrations are elevated relative to reference area data and due to its contributions to risk as follows:
  - Cu concentrations in porewater, along with other SEM concentrations, contribute to an exceedance of their summed benchmarks (which are protective of benthic organisms and fish) in some portions of the Study Area.
  - In addition, the spatial pattern for Cu in surface sediment is generally similar to that for other metals (see Attachment A-A1 of Appendix A), indicating it is a representative metal.

- Pb was identified in the BERA as a COPEC based on dietary intake for the spotted sandpiper, primarily from the incidental ingestion of sediment while foraging.
- While dieldrin was not identified as a COPC or COPEC in the BHHRA or the BERA, respectively, spatial evaluation of tissue data for polychaetes shows that dieldrin tissue concentrations in English Kills exceed the lowest observed effect concentration (LOEC) provided by USEPA for benthic macroinvertebrates. Therefore, dieldrin was evaluated further in the nature and extent of surface sediment and tissue but not in other media.

Treatment of the data (e.g., methods for querying, processing, and calculations of quantities from the data, such as summations) is discussed in Section 4.1.3, and a description of the presentation tools used to evaluate the nature and extent of contamination for these eight contaminants (or groups of contaminants) is provided in Section 4.1.4. The nature and extent of contamination for each sampling medium (i.e., surface sediment, subsurface sediment, native material, sediment traps, surface water, porewater, groundwater, and tissue) is presented in Sections 4.2 through 4.10, for each of these eight contaminants (or groups of contaminants), with the following exceptions:

- Dieldrin is only discussed for surface sediment and tissue (based on the risk assessments).
- 2,3,7,8-TCDD is not discussed for porewater or groundwater, because those media were not analyzed for that chemical.
- C19-C36 is not discussed for surface water, shallow porewater, or tissue, because those media were not analyzed for that chemical.
- TPAH (34) is not discussed for tissue, because PAH analysis of that medium was based on TPAH (17).

For ambient air, a characterization of environmental conditions based on VOCs and TPCB is presented in Section 4.11.

Sections 4.2 through 4.10 include key figures and tables relating to the nature and extent of the eight contaminants (or groups of contaminants); figures and tables for all contaminants (including these eight), as well as conventional parameters, are provided in Attachment A-A of Appendix A.

Comparisons between characteristics and distributions of these eight contaminants (or groups of contaminants) by chemical class (i.e., hydrocarbons, bioaccumulative organics, and metals) is provided in Section 4.12, including a discussion of which of these contaminants are evaluated further for sources and fate and transport in Sections 5 and 6 of this report.

4.1.3 Data Treatment Used in the Analysis

This section provides details regarding the application of the RI data specific to the evaluation of nature and extent.

4.1.3.1 TPCB in Surface Sediment, Subsurface Sediment, and Native Material

Surface and subsurface sediment samples and native material samples collected in Phase 1 and as part of the National Grid (2009/2010) sampling were analyzed for PCBs in accordance with USEPA Method 8082 ("Aroclor" data). Twenty-five percent of the Phase 1 samples, as well as all Phase 2 samples, were analyzed in accordance with USEPA Method 1668 ("congener" data).[71]

The Aroclor and congener datasets were combined to create a single dataset for TPCB.[72] A strong linear correlation (coefficient of determination [r^2] = 0.87) exists between the two analytical methods, as demonstrated using the paired TPCB congener and TPCB Aroclor surface and subsurface sediment and native material data from the Phase 1 dataset (see Figure 4-1[73]). To create a single unified dataset, the TPCB Aroclor data were multiplied by the average ratio of TPCB congener to TPCB Aroclor concentration in these paired samples of 1.75 (this value is represented by the dashed line in Figure 4-1) to provide a concentration of

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[71] See Section 4.2.5.2 for a discussion of PCB composition.

[72] The use of the combined dataset was approved in an e-mail from USEPA to Anchor QEA on April 5, 2016 (Kwan 2016a). Because some of the individual congeners and Aroclors were not detected in some of the samples, for both analytical methods, the Kaplan-Meier (KM) method was used to treat non-detect individual congeners or Aroclors when summing to calculate TPCB. If, following summation, TPCB was non-detect, TPCB was set to the method detection limit (MDL). See Section 3.2.4.2 of the BHHRA (see Appendix H) and Attachment D6 of the BHHRA for a discussion of the KM method.

[73] Figure 4-1 includes surface sediment, subsurface sediment, and native material samples to produce a relationship for use for all three media throughout this RI Report.

TPCB congeners.[74] This combined dataset is used in the evaluation of TPCB in sediment throughout this RI Report.

4.1.3.2 TPAH (17) and TPAH (34)

TPAH (17) was calculated as the sum[75] of the following compounds:

- 2-methylnaphthalene
- Acenaphthene
- Acenaphthylene
- Anthracene
- Fluorene
- Naphthalene
- Phenanthrene
- Fluoranthene
- Pyrene
- Benzo(a)anthracene
- Chrysene
- Benzo(b)fluoranthene
- Benzo(j,k)fluoranthene[76]
- Benzo(a)pyrene
- Indeno(1,2,3-c,d)pyrene
- Dibenzo(a,h)anthracene
- Benzo(g,h,i)perylene

These include the 16 USEPA priority pollutant PAHs, as well as 2-methylnaphthalene. TPAH (34) was calculated as the sum of the following compounds:

- Acenaphthene
- Acenaphthylene

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[74] The National Grid TPCB data (also Aroclor) were multiplied by the same factor, on the basis that the same analytical method and laboratory were used in both the Phase 1 and National Grid sampling, and no significant changes were made to relevant analytical procedures at that laboratory in the time between the National Grid program (2010) and the Phase 1 program (2012).

[75] The sum was calculated using the KM method, with non-detects for the summed total set equal to the MDL.

[76] These two compounds cannot be resolved analytically, so the reported concentration is for the total of the two.

- Anthracene
- Benzo(a)anthracene
- Benzo(a)pyrene
- Benzo(b)fluoranthene
- Benzo(e)pyrene
- Benzo(g,h,i)perylene
- Benzo(j,k)fluoranthene
- Chrysene
- Dibenzo(a,h)anthracene
- Fluoranthene
- Fluorene
- Indeno(1,2,3-c,d)pyrene
- Naphthalene
- Perylene
- Phenanthrene
- Pyrene
- C1-Naphthalenes
- C1-Fluorenes
- C1-Benzanthracenes/Chrysenes
- C1-Fluoranthenes/Pyrenes
- C1-Phenanthrenes/Anthracenes
- C2-Benzanthracenes/Chrysenes
- C2-Fluorenes
- C2-Naphthalenes
- C2-Phenanthrenes/Anthracenes
- C3-Benzanthracenes/Chrysenes
- C3-Fluorenes
- C3-Naphthalenes
- C3-Phenanthrenes/Anthracenes
- C4-Benzanthracenes/Chrysenes
- C4-Naphthalenes
- C4-Phenanthrenes/Anthracenes

TPAH (34) includes both the 17 compounds in TPAH (17), as well as 17 other C1- to C4-alkylated homologs of 2- to 6-ring PAHs. Additional discussion of these PAH sums is provided in Section 4.2.5.1.

4.1.3.3 Total Organic Carbon

TOC was measured in every surface and subsurface sediment sample, as well as in native material, both in the RI datasets and in the National Grid dataset. The Phase 2 data and the National Grid data satisfied all quality control requirements and were used in the RI datasets. The TOC results from the Phase 1 sediment samples exhibited a low bias, due to laboratory error (see Table 2-2b and Appendix Biii [see Table Biii1-2]), so these samples were rejected. In consultation with and with the approval of USEPA, available Phase 1 archived samples were reanalyzed. The reanalyses satisfied all quality control evaluations (Phase 2 QAPP Deviation Form 9-6). However, archived samples were available for only 559 of the 793 Phase 1 samples.

Based on discussion with USEPA (Kwan 2016b), a multiplier was developed using the 559 paired original and reanalyzed Phase 1 archived samples and then applied to the 234 original Phase 1 TOC data for which archived samples were not available. The multiplier was calculated using a linear regression between the original and reanalyzed Phase 1 TOC data (with intercept fixed at zero; a slope of 1.56 was calculated; see Figure 4-2). Those data in the subset of original Phase 1 TOC data for which archived samples were not available were multiplied by 1.56 to obtain a corrected TOC value to include in the RI dataset. In this way, a single unified RI dataset was created for TOC. Due to the uncertainties associated with the original Phase 1 data, analyses in this report that involve TOC were performed both with and without these corrected original Phase 1 TOC data, as a sensitivity analysis, and any cases where results or conclusions changed are noted.

4.1.3.4 Definition of Surface Sediment

Surface sediment samples from the RI and Part 1 of the FS field program were collected as grab samples from the top 15 cm (6 inches) of the sediment bed (with a few exceptions[77]; see Section 4.1 of Appendix Bi and Section 5.3 of Appendix Bii for more discussion of the sampling methods). All National Grid surface sediment samples were collected as grab samples from the top 10 cm (4 inches), and collected cores were collocated with the grab locations. For those locations where the collocated core was analyzed, a weighted average chemical concentration was developed based on analytical results of the 0- to 10-cm (0- to 4-inch) and 10- to 20-cm (4- to 8-inch) National Grid sediment samples to characterize the 15-cm surface sediment interval.[78]

4.1.3.5 Surface Water Particulate Phase Concentrations

In order to compare concentrations of contaminants that are bound to particulates in surface water with concentrations in other environmental media (such as sediment), the particulate phase concentration is calculated. The particulate phase concentration is the mass of a chemical per unit mass of dried suspended sediment from a water sample. The particulate phase concentration can be determined by comparing direct measurements of chemical concentrations in whole water (with particulates, also termed “unfiltered”) and filtered water (without particulates), or it can be estimated using partitioning theory. Direct measurement of whole

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[77] When a full 15-cm (6-inch) grab could not be collected (e.g., due to obstructions), the RI/FS Work Plan (AECOM 2011) and Phase 2 RI Work Plan Volumes 1 and 2 (Anchor QEA 2014a, 2014b) called for multiple attempts. After three attempts, the grab was to be accepted if the depth was greater than 10 cm (4 inches). In 54 of 510 surface sediment grab samples, depths were between 6 and 15 cm (2 and 6 inches; only 6 samples were shallower than 10 cm [4 inches]). Two Phase 2 programs collected surface sediment samples from the top 15 cm (6 inches) of cores—the NYC post-dredge area sediment sampling (see Section 4.2.4) and the groundwater program (see Section 2). One of these 27 samples was from the top 13 cm (5 inches). The surface sediment samples collected for the shoreline portion of Part 1 of the FS field program were collected from the top 15 cm (6 inches) of sediment.

[78] For 22 of the 31 National Grid cores, a segment spanning the depth 10 to 20 cm (4 to 8 inches) was collected and analyzed (deeper segments were analyzed as well) in addition to the 0- to 10-cm (0- to 4-inch) interval analyzed at all locations. Based on discussion with USEPA (Kwan 2016a), an equivalent 0- to 15-cm (0- to 6-inch) sample was created mathematically from these data using a weighted average, employing the following formula: 0.67 × [concentration in the 0- to 10-cm (0- to 4-inch) grab] + 0.33 × [concentration in the 10- to 20-cm (4- to 8-inch) core segment]. For the 22 samples with both of these intervals analyzed, this weighted average was used to represent surface sediment; for the remaining 9 grab samples, the result from the 0- to 10-cm (0- to 4-inch) grab was used.

water and filtered water samples is used to calculate particulate phase concentration for some analytes, whereas partitioning theory is used when only whole-water samples were collected for a given analyte. The sampling conducted in support of this RI Report used both techniques.

For metals, the surface water sampling program included collection of both whole-water and field-filtered dissolved phase samples. For these analytes, particulate phase concentrations were calculated by subtracting the dissolved phase results from the whole-water results and dividing the resulting difference by the TSS concentration to produce a particulate phase metals concentration on a mass per solids (e.g., mg/kg dry weight) basis.

For organics, including PAHs and PCBs, only whole-water samples were collected. As a result, the particulate phase concentrations of PAHs, PCBs, and pesticides were estimated based on partitioning theory using site-specific partition coefficients (see Section 6.4.1) along with the measured TSS concentration of each sample. Partitioning relationships for additional chemicals are displayed in figures provided in Attachment A-C of Appendix A. A more detailed description of the calculations used to generate estimates of particulate phase concentrations is presented in Attachment E-C of Appendix E.

4.1.3.6 Tissue Data

Tissue samples were analyzed as whole-body composites or in composites created using specific tissue types (e.g., fillet and carcass) to meet the needs of the risk assessments (see Appendices H and I). Data are presented in this report for the following tissues:

- Striped bass: fillet and whole body
- White perch: fillet
- Atlantic menhaden: whole body
- Mummichog: whole body
- Blue crab: muscle+hepatopancreas and whole body
- Bivalves: whole body
- Benthic invertebrates (polychaetes): whole body

All tissue samples were analyzed as composites of multiple individual organisms to provide sufficient tissue mass for all required chemical analyses, and in the case of fish and crab, to efficiently increase the number of individuals included in the dataset to improve the reliability of the data. Fish and crab composites consisted of individuals of similar size, using the target rule that the smallest individual in a composite was no less than 75% of the total length or carapace width of the largest individual, respectively.

For striped bass, some composites were analyzed as whole-body samples for the BERA (see Section 4.2.5 of Appendix I) and some as fillets for the BHHRA (see Section 3.1.2.2.3 of Appendix H). For efficiency, the remaining carcasses from the fillet composites were also analyzed (as composites), and then using the matched fillet and carcass data, equivalent whole-body concentrations were calculated, as described in the BERA (see Section 4.3.4.4 of Appendix I). Similarly, some blue crab composites were analyzed as whole-body samples (BERA), and some were analyzed as separate muscle tissue, hepatopancreas tissue, and carcass. For the human health risk assessment, reconstituted muscle plus hepatopancreas concentrations were calculated as described in Section 3 of the BHHRA (see Section 3.2.4.4 of Appendix H). For the ecological risk assessment, equivalent whole-body concentrations were calculated as described in Section 4 of the BERA (see Section 4.3.4.4 of Appendix I).

4.1.4 Description of Presentation Tools

The nature and extent evaluations are presented using five primary types of graphics and tables of summary statistics, which are described in this subsection. In addition to these, other types of graphics are used, as appropriate, for specific media and are described in the appropriate places in this document. Figures and tables containing results for the full set of chemical analytes for each media are provided in Attachment A-A of Appendix A. The five primary types of graphics are briefly described as follows:

- **Plan view maps** show data at individual sampling locations. Their primary purpose is to show the spatial patterns of the data. Therefore, quantiles of the data are used to facilitate visual observation of spatial patterns. Data values at each location are presented as color-coded symbols; each color represents a specific range of concentrations, based on quantiles of the data (e.g., 20th, 40th, 60th, and 80th percentiles). In most cases, quintiles are used; however, quartiles are sometimes used when there is a small number of samples or unique values. In some cases, these quantiles were modified to provide rounded values. For presentation of subsurface sediment data, plan view maps showing stacked bars color coded by concentration ranges are used to display concentration by depth at each core location.

- **Longitudinal profiles** show individual chemical concentrations throughout the Study Area. Their primary purpose is to show the spatial patterns of the data as a function of distance from the confluence of Newtown Creek with the East River. For comparison, the concentrations from the Phase 1 and Phase 2 reference areas are shown in these figures, in most cases. Concentrations are presented on the vertical axis, versus the CM on the horizontal axis; CM 0.0 is the mouth of the creek at the confluence with the East River. Therefore, a sample in a tributary and a sample in the main stem may have the same CM, even though they are separated in space. The main stem and the tributary samples are indicated by different colors to allow them to be distinguished from one another.[79] Figure 4-3 presents the CM system used in the Study Area. Reference area data are shown on the right side of the Study Area data, separated by a vertical line. Data for each reference area are shown as a vertical stack of points, rather than as a function of CM in the reference area, because the primary purpose of including the reference area data is to show the range of concentrations in each reference area compared to the Study Area.
- **Box plots** present the data grouped according to reach within the Study Area (based on individual tributaries and CM in the main stem, as discussed in Section 4.1) and the Phase 1 and Phase 2 reference areas. Their primary purpose is to show the distribution of data within a Study Area reach or a reference area and facilitate comparison among reaches and between the Study Area and reference areas. The boxes span the 25th and 75th percentiles of the data (i.e., the interquartile range). The horizontal line through each box indicates the median. Whiskers extend beyond the boxes to the 10th and 90th percentiles. All individual data values beyond the 10th and 90th percentiles are presented as individual symbols.[80] The coloring scheme of the boxes matches the coloring of the longitudinal profiles. Sample counts are posted at the top of the panel.

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[79] In the case of surface sediment data, a variation of the longitudinal profile is used to allow for a more detailed view of spatial patterns. Each figure is presented on two pages: on the first page, the main stem and English Kills are shown, and on the second, the other tributaries are shown on separate plot panels.

[80] For sample sets with less than or equal to 10 values, whiskers are not provided, and all data lying outside the interquartile range are presented as individual symbols. For sample sets with less than five values, boxes are not presented, and all data are presented as symbols.

The boxes present the data from each tributary individually and the main stem for CM 0 – 1, CM 1 – 2, and CM 2+ (which includes the Turning Basin).[81]

- **Cross plots** present a comparison of two sets of data (e.g., surface versus subsurface sediment chemical concentrations within a core, or one analyte or location versus another) whose purpose is to allow for a visual comparison of the relationship between the two sets of data, including whether one is systematically higher or lower than the other. In cases where cross plots are used to evaluate systematic differences between two sets of data (e.g., to evaluate whether surface or subsurface concentrations are generally higher within sediment cores), a 1:1 line is plotted to visually identify the position of each dataset relative to the other. For such cross plots, a binomial statistical test is included to test the null hypothesis that the two datasets are equally distributed above and below the 1:1 line and are not systematically different. If the significance level (p value) from the binomial test is < 0.05, the null hypothesis is rejected, indicating that one of the two datasets is systematically higher than the other. In other cases where cross plots are used and a meaningful functional relationship may be expected between two sets of data (e.g., TPCB congener versus TPCB Aroclor for samples analyzed by both methods), a regression line (typically a linear regression line, but in some cases the regression is performed in log space) and corresponding r^2 value are shown.
- **Vertical profiles** present data by depth in the sediment bed in the Study Area, with depth on the vertical axis (zero defined as the surface of the sediment bed) and the data value on the horizontal axis. These profiles may represent individual locations (e.g., a single sediment core location) or groupings of locations (e.g., all sediment cores from a given reach). For the latter case, vertical profiles are shown using box plots, in which samples are grouped into panels by reach (tributary and main stem separately). Depth intervals for the boxes are based on the generally used subsurface sampling intervals in the Phase 2 RI Work Plan Volume 1 (Anchor QEA 2014a). For each box,

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[81] Discussions of spatial patterns of the data in this section are organized by reach using spatial maps, longitudinal profiles, and box plots. This approach allows for a systematic description of the distributions of concentration to define nature and extent. Discussions are not centered around proximity to upland sites or other potential sources, because to do so would be speculative. Chemicals in the Study Area cannot necessarily be attributed to proximate upland site(s) or source(s), including point sources, because of the complex patterns of development and use of upland sites, the complex history of sediment deposition and transport in the Study Area, dredging and modifications to the course of the creek over time, and a lack of upland data for many sites.

all core segments that include the specified depth are included (i.e., no length weighting was performed). For example, all core samples with a depth interval that includes 80 cm are grouped into a box plotted at a depth of 80 cm. These box plots allow for comparison of the central tendency and distribution of data values within each area by depth in a simple manner.

In addition to these various graphics, tables presenting summary statistics by reach are also provided for each media and chemical discussed. In the text, concentration ranges and averages are provided in summary bullets by reach in the "Distribution of Contaminants" subsections throughout Section 4. As discussed in the General Table, Figure, and Chemistry Data Notes (after Section 10), for chemicals that have fewer than two detected results in a given reach, the minimum and maximum results are shown in the tables, and the remaining statistics were not calculated, because doing so would be meaningless. In such cases, the summary bullets listed in the remainder of this section present a concentration range but no arithmetic average (with the minimum value representing a non-detect).

4.2 Surface Sediment

4.2.1 Surface Sediment Dataset

The surface sediment dataset includes samples that were collected to a depth of 15 cm (6 inches), with a few exceptions noted in Section 4.1.3.4. As described in Sections 4.1.1 and 4.1.3.4, the dataset includes samples from all Phase 1 and Phase 2 surface sediment sampling programs and the shoreline sediment sampling from Part 1 of the FS field program (see Table 2-2b), as well as the National Grid dataset. As described in Section 3.2.4, maintenance dredging was performed by NYCDEP during April and May 2014, in CM 0 – 1 and Whale Creek in the time between completion of the Phase 1 and Phase 2 sampling programs. Phase 1 data (three samples) that were collected within the dredging footprint were removed from the dataset to ensure that the dataset represents current conditions. Phase 2 data that were collected within the dredging footprint (10 samples) are used to define surface concentrations in those areas (see Table 4-10). A discussion of the impact of that navigational dredging on surface sediment concentrations is provided in Section 4.2.4.

4.2.2 Percent Fines and Total Organic Carbon

Percent fines and TOC are key characteristics of surface sediment, subsurface sediment, and native material that provide important insights regarding sediment transport and CFT, which in turn affect chemical distribution patterns.

The presence of OC compounds in surface sediment, subsurface sediment, and native material is an important factor to consider in the RI, because the process of sorption of contaminants to these media is controlled to a large degree by the amount of OC present, which reduces the bioavailability of these contaminants. For example, contaminants that are more strongly sorbed to OC compounds tend to be less abundant in porewater and surface water, meaning that they are not as bioavailable and less likely to impact benthic organisms that take up dissolved contaminants from these media. As well as naturally occurring OC compounds, OC compounds in urban settings also include anthropogenically derived materials, including soot and petroleum products. This complicates this behavior, because the OC consists, in part, of contaminants that are the subject of this nature and extent evaluation (e.g., TPAH [17], TPAH [34], and C19-C36).

The distribution of grain sizes and organic matter in surface and subsurface sediment depends not only on the locations of their sources, but also on circulation and depositional characteristics of the system. Newtown Creek has no remaining natural fresh surface water inputs, which influences the distribution of fine sediment and OC. There are sources of organic matter and solids within the tributaries (i.e., CSOs, MS4s, and large stormwater outfalls, as subsequently discussed in Section 5.1 [e.g., see Figure 5-14]), as well as at the downstream boundary at the East River. There are also inputs of organic matter and solids from individually permitted stormwater and wastewater discharges, as well as overland flow. In addition, OC in the surface and subsurface sediment reflects inputs from shoreline seeps (where present) and historical spills and releases, as discussed in Section 3.2.6. The location and influence of these sources of organic matter vary spatially within the Study Area.

Contaminants sorb to OC, which means that the distribution of contaminants within the Study Area is influenced not only by the sources of the contaminants, but also by the sources and fate and transport of solids and organic matter, which is complex in Newtown Creek. This section describes the distribution of grain size and organic matter content, with the goal of supporting the understanding of the distribution of contaminants, which is discussed in subsequent portions of Section 4, as well as Sections 6 and 8.

4.2.2.1 Percent Fines

The surface sediment in the Study Area is generally fine-grained, consistent with the low-energy, depositional nature of the system (see Section 3.1.2). The percent fines in the surface sediment of the Study Area (i.e., the relative amount of clay- and silt-size particles, which consist of particles less than 62.5 micrometers in diameter) exceed 40 wt% in 76% of the samples (see Table 4-11 and Figures 4-4 through 4-6). There is considerable overlap in the percent fines data among the reaches of the Study Area, between the Study Area and the reference areas, and among the reference area categories. The highest percent fines are generally found in the main stem below approximately CM 2, the downstream portion of Dutch Kills (approximately CM 1), and the western half of the Turning Basin (approximately CM 2.5). Although there is variability in the data in all reaches, somewhat lower percent fines values are generally found in the tributaries. The arithmetic average percent fines in tributaries with CSOs (i.e., Dutch Kills, Maspeth Creek, East Branch, and English Kills) ranges from 43 wt% to 57 wt% (medians range from 40 wt% to 64 wt%), which is consistently lower than in the main stem reaches. Arithmetic averages in the main stem range from 64 wt% to 82 wt%, and medians range from 72 wt% to 90 wt% (see Table 4-11). Whale Creek has an arithmetic average and median percent fines closer to the main stem range (63 wt% and 71 wt%, respectively). CSO effluent contains both coarse-grained and fine-grained material, with percent fines in CSO point source samples ranging from 13 wt% to 91 wt% (arithmetic average 49 wt%).[82] The rapid settling of coarse-grained material compared with fine-grained material provides an explanation for the general finding of somewhat lower percent fines in sediments in the tributaries relative to the main stem, recognizing that there is local variability within all of

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[82] The grain size distribution of suspended sediment material in the CSO point source samples was analyzed using two analytical methods: ASTM International (ASTM) D3977 (which quantified fine and coarse suspended sediment concentrations) and a laser diffraction-based method (see Table B10-4 from the Phase 2 FSAP Volume 2 [Anchor QEA 2014d]). The data from the laser diffraction-based method generally showed lower percent fines relative to those from the ASTM D3977 method. The statistics presented in the text reflect the range of percent fines calculated using the results of the laser diffraction-based method. Using the ASTM D3977 method, the arithmetic average percent fines in CSO samples ranges from 23 wt% to 100 wt%, with an arithmetic average of 69 wt%. The difference in arithmetic average percent fines between the two methods does not change the conclusion that there is a portion of the CSO suspended sediment material that is coarse-grained, and this portion will still settle faster than the fine-grained portion.

the Study Area reaches. Additional evaluation of percent fines and grain size is discussed as part of the sediment transport modeling documented in Section 5.2.3 of Appendix G.

4.2.2.2 Total Organic Carbon

The arithmetic average for surface sediment TOC content in the various Study Area reaches ranges from 3.9 wt% to 12 wt% (median values range from 3.8 wt% to 12 wt%; see Table 4-12 and Figures 4-7 through 4-9). Lower TOC concentrations are found in the main stem below CM 2 (range of less than 1 wt% to 10 wt%, with an arithmetic average of 3.9 wt% and median of 3.8 wt%) and Whale Creek (arithmetic average of 4.6 wt% and median of 5.0 wt%). Higher concentrations are found in CM 2+ (range of less than 1 wt% to 32 wt%, with an arithmetic average and median of 9.2 wt%). The highest arithmetic average TOC concentrations are generally found in the four tributaries that contain CSOs, with arithmetic average TOC concentrations ranging from 9.0 wt% to 12 wt% (median values range from 8.6 wt% to 12 wt%; see Figures 4-8 and 4-9 and Table 4-12).[83] The elevated TOC found in these tributaries is likely due to a combination of releases from CSOs as well as industrial sources and stormwater.

TOC in the Study Area tributaries is high relative to the overall TOC content of New York Harbor sediments. In the New York/New Jersey (NY/NJ) Harbor Regional Environmental Monitoring and Assessment Program (REMAP) study, the mean TOC in Harbor sediments was 2.3 wt%, and TOC means in the subbasins ranged from 1.7 wt% to 3.8 wt% (USEPA 2003a). Similarly, TOC levels in the Study Area generally exceed those in the reference areas. The Phase 1 and Phase 2 reference areas, some of which include CSOs, exhibit arithmetic average TOC levels that range from 1.8 wt% to 9.2 wt%.

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[83] Dutch Kills is one of these tributaries. Sediment transport modeling shows current sedimentation rates in Dutch Kills are driven more by solids inputs from the East River than from point sources (e.g., see Section 5.6.1.1 of Appendix G). TOC from the East River is lower than that from CSOs (see Sections 5.3.1 and 5.1.3.3.2, respectively), but Dutch Kills surface sediment TOC is relatively high. This higher TOC observed in Dutch Kills surface sediment as compared to the main stem and East River could be a result of: 1) additional OC inputs to Dutch Kills (beyond CSOs and East River); and/or 2) potential changes in sources of OC and solids over time. Additional OC inputs include industrial point sources and shoreline seeps, where present. With respect to changes in sources of OC and solids over time, current contributions to net sedimentation are likely different than they were historically. Historical contributions of OC and solids likely changed as a result of infrastructure changes and changes in industrial and municipal activities and discharges, as discussed in Section 3.2. Given surface sediments reflect a blend of recent and historical influences (as discussed in Section 8.6.1.4), the current surface sediment OC may not be equal to that of the depositing solids.

The reference area TOC levels vary considerably, with arithmetic average TOC values ranging from values that are similar to those in CM 0 – 2 (i.e., around 5 wt%) to values that are similar to the highest values seen in the tributaries (i.e., around 10 wt%). The ranges of TOC in reference areas (see Section 2.1.2) classified as Industrial compared to Non-Industrial are generally comparable. The arithmetic average TOC in Industrial reference areas ranges from 1.8 wt% to 9.2 wt%, while those in Non-Industrial reference areas range from 2.5 wt% to 9.0 wt%. Similarly, the ranges of TOC levels in reference areas with CSOs are generally similar to those of reference areas without CSOs. The arithmetic average TOC in reference areas with CSOs is 2.5 wt% to 9.2 wt%, while the range in reference areas without CSOs is 1.8 wt% to 6.5 wt%. Two reference areas (Coney Island Creek and Fresh Creek) exhibit arithmetic average TOC levels approaching those of the four Study Area tributaries, with CSOs (arithmetic averages ranging from 9.0 wt% to 12 wt%; see Table 4-12 and Figure 4-9). Coney Island Creek is classified as Industrial/CSO and has an arithmetic average TOC level of 9.2%. Fresh Creek is classified as Non-Industrial/CSO (although it does have some degree of industrial activity; it is an SMIA) and has an average TOC level of 9.0%, similar to Coney Island Creek. These results suggest that industrial sources and CSOs contribute organic matter to New York Harbor tributaries, including the Study Area.

Finally, the upper ends of the CSO-containing tributaries in Newtown Creek are characterized by both relatively high TOC and low percent fines. The combination of high TOC and low percent fines runs counter to the usual expectation of a positive correlation between percent fines and TOC in aquatic sediments.[84] The finding that CSO effluent contains high TOC concentrations and significant proportions of larger particles (see Section 4.2.1.3 of Appendix E) points to the influence of the major CSOs on these sediment properties in the tributaries. Industrial sources and stormwater also have contributed some OC to these areas. The relationship between percent fines and TOC is discussed further in Section 4.2.2.3.

4.2.2.3 Relationship Between Percent Fines and Total Organic Carbon

In aquatic systems, percent fines and TOC usually exhibit a positive relationship in surface sediment, as fine material tends to be richer in organic matter (see discussion in

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[84] For example, the REMAP program (USEPA 2003a) included an extensive surface sediment sampling program in the NY/NJ Harbor. A comparison of regional average percent silt/clay (see Table 3-2 of USEPA 2003a) and TOC (see Table 3-3 of USEPA 2003a) shows a positive relationship (r = 0.86).

Section 4.2.2.2). The Newtown Creek system is unusual, due to the presence of more than one significant source of solids (East River, point sources) with different particle size distributions and compositions. Furthermore, there are multiple sources of TOC, including the East River, point sources within Newtown Creek, historical spills and releases, and other sources, as discussed in Section 3.2.6. The relationship between percent fines and TOC was evaluated in both the Study Area and the Phase 1 and Phase 2 reference areas in order to better understand the nature and sources of sediment and OC depositing in the Study Area.

The relationship between arithmetic average percent fines and arithmetic average TOC in surface sediment from the Study Area is shown in Figure 4-10a. Within the Study Area, surface sediment percent fines and TOC were averaged in 0.5-mile increments, with each tributary and each reach of the main stem averaged separately. The surface sediment data from the main stem and East River reference areas (see Figure 4-10b) are also included (includes "Lower East River – Newtown Creek" and "Upper East River – Throgs Neck" reference areas; averaged in 0.5-mile increments), because this waterbody influences the lower portion of the Study Area. Finally, for comparison, point source particulate phase data are included, averaged by type of point source (see Section 5.1.1). These data represent the other key sources of solids to the Study Area.

The TOC versus percent fines data are clustered, as indicated in Figure 4-10b. East River surface sediment (circled in gray in Figure 4-10b) exhibits lower percent fines (5 wt% to 60 wt%)[85] and lower TOC (less than 1 wt% to 4 wt%) than the other datasets. Surface sediment from the lower portion of the Study Area (CM 0 – 1, CM 1 – 2, and Whale Creek; circled in orange in Figure 4-10b) cluster in line with the East River data, although with higher percent fines (60% to 90%) and higher TOC (3 wt% to 6 wt%) than the East River. These datasets together (i.e., East River and Study Area CM 0 – 1, CM 1 – 2, and Whale Creek) exhibit a positive relationship between percent fines and TOC ($n = 11$, correlation coefficient [r] = 0.91, $p < 0.05$), as would be expected. This is consistent with the other analyses presented in this RI Report, as well as the sediment transport model, which indicate the solids depositing in the lower portion of the Study Area originate

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[85] The ranges of TOC and percent fines presented in the following paragraphs reflect the 0.5-mile averaged values for surface sediment and the point source data averaged by category as shown in Figure 4-10b.

primarily from the East River (see Section 5.6.1.2 of Appendix G). Finer solids and higher TOC are expected in the Study Area sediment due to its relatively quiescent nature, compared with the East River.

The Study Area point source data (circled in red in Figure 4-10b) exhibit intermediate values for percent fines (ranging from 45 wt% to 67 wt%) and the highest TOC of the data included in this analysis (13 wt% to 41 wt%) (see Section 5.1 of this RI Report and also Section 2.3 of Appendix E for more discussion of point sources data).[86] The surface sediment in the Study Area tributaries (which plot in the center of Figure 4-10b) exhibits similar percent fines to the point sources (40 wt% to 60 wt%) and lower TOC than the point sources (9 wt% to 12 wt%). This is consistent with the loss of organic matter relatively soon after deposition of point source solids in the tributaries. The data from CM 2+ are intermediate between the tributaries and CM 0 – 2, supporting a mixing process between East River and upstream sources of solids and OC, consistent with the sediment transport CSM (see Section 6.3).

Finally, historically deposited industrial and municipal compounds are also sources of the OC presently found in the Study Area, and some portion of the OC present in surface sediments can be a remnant of these historical sources, due to mixing within the surface sediment (see Section 6.4.4.4), particularly in the tributaries and CM 2+ (which exhibit higher TOC than CM 0 – 2).

4.2.2.4 Total Organic Carbon Composition

OC in the surface sediment of the Study Area consists of a complex mixture of compounds, originating from both natural and anthropogenic sources. Sources include historical and ongoing discharges of wastewater, urban runoff from stormwater outfalls and CSOs, historical spills and releases, shoreline seeps, and other sources, as discussed in Sections 3.2.6 and 4.2.2.3. A portion of this OC is labile (i.e., readily degraded by bacteria), as evidenced by the low DO concentrations in the surface water (often less than 3 mg/L) and the consequent need for

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[86] TOC from point sources in this context is equivalent to fraction organic carbon (f_{OC}) = particulate organic carbon (POC)/TSS (see Section 5.1.3.3.2). The percent fines data for point source samples shown in this figure are based on the laser diffraction-based analytical method. Values based on the ASTM D3977 method are somewhat higher (see Section 4.2.2.2 and Footnote 82), which would cause the data to plot farther to the right on the figure, but would not change the conclusion that the Study Area sediment reflect a mix of sources of solids.

surface water aeration (see Section 3.2.8.3.1). However, much of the OC is more refractory (i.e., less readily degraded by bacteria or completely resistant to degradation) and remains in the sediment bed for long periods of time (at least decades), as evidenced by the elevated TOC levels observed in subsurface sediment (see Section 4.3). Both "modern" carbon (i.e., carbon from organic material that was recently synthesized through biological processes; this includes carbon that continues to be deposited in the system from current discharges, as well as carbon that has been deposited within the last century or so) and "old" carbon (i.e., carbon that originated from material that was living millions of years ago) can include relatively labile and recalcitrant components. As mentioned previously, low DO in the water suggests that a significant fraction of ongoing sources of organic matter are labile. In addition, the high proportion of HPAH in Newtown Creek sediments (see Section 4.2.5.1) points to reduced lability of "old" carbon (Heitkamp and Cerniglia 1987). The relative overall lability of organic matter from ongoing and historical sources has not been quantified for Newtown Creek. The rest of this section describes the composition of the more refractory OC in the Study Area.

During the RI, several analytical measurements were performed to provide information on the composition of OC (measured as TOC) in surface sediment; these measurements include TPH (C9-C40), C5-C8 aliphatics, C9-C12 aliphatics, C9-C18 aliphatics, C19-C36 aliphatics, C9-C10 aromatics, C11-C22 aromatics, DRO (C10-C28), and soot carbon (highly condensed carbonaceous residue from incomplete combustion processes).[87] The focus of this subsection is on TPH (C9-C40) and soot carbon, for the reasons discussed in the following paragraphs.

TPH is an analytical parameter that includes a complex mixture of thousands of individual aromatic and aliphatic compounds,[88] from a variety of sources, and constitutes a portion of TOC. TPH concentrations in the Study Area generally range from 10^3 to 10^5 mg/kg (0.1 wt% to 10 wt% of dry sediment mass) in surface sediment. Concentrations in the main stem increase upstream from the mouth to approximately CM 2.8. Concentrations in Dutch Kills and Whale Creek are elevated relative to the adjacent portion of the main stem, whereas concentrations in

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[87] Phase 1 data measured TPH (C9-C40) and DRO (C10-C28) in sediment samples. Other smaller ranges of measurements of TPH (e.g., C5-C8 aliphatics) were measured only for the Phase 2 program. The National Grid dataset does not include TPH measurements.

[88] The compounds included in TPAH (17), TPAH (34), C19-C36, and DRO are all included in TPH. A comparative evaluation of these various hydrocarbon sums in surface sediment is presented in Section 4.12.1.1.

Maspeth Creek, English Kills, and East Branch are similar to the main stem concentrations in the vicinity of CM 2.8 (see Figure 4-11). The arithmetic average of the ratio of TPH to TOC is 0.13 in surface sediment, ranging generally from 0.05 to 0.50 (see Figure 4-12).

Soot carbon concentrations range from approximately 0.1 wt% to 2 wt% of dry matter in the Study Area surface sediment (see Figure 4-13). In general, soot carbon is spread fairly evenly throughout the Study Area, with a few high values scattered along the main stem, including CM 2+ (see Figure 4-13). The arithmetic average of the ratio of soot carbon to TOC is 0.08 in surface sediment, ranging generally from 0.01 to 0.20, with a few values near 1.0 (see Figure 4-14).

Different types of OC sorb chemicals to differing degrees. For example, soot carbon is generally known to exhibit strong sorption of organic compounds compared with other forms of organic matter found in nature (Jonker and Koelmans 2002). The various fractions of TOC listed previously in this subsection vary in effects on partitioning, so the overall partitioning behavior of the mixture that comprises the TOC can become quite variable. This is important for both the RI and modeling, because of the important role that TOC plays in understanding sorption processes. TOC is often used as a normalizing component in understanding contaminant distribution patterns, partitioning, and bioaccumulation. Contaminant concentrations are often expressed in units of mg/kg OC (which equals the ratio of dry weight-based chemical concentrations and TOC), as shown in the following equation:

$$\left[\frac{\text{mg contaminant}}{\text{kg dry weight}}\right] / \left[\frac{\text{kg OC}}{\text{kg dry weight}}\right] = \left[\frac{\text{mg contaminant}}{\text{kg OC}}\right] \tag{4-1}$$

Due to the complex nature of TOC composition in this system and the variability in TOC composition, it is not clear that simple carbon normalization (as this ratio approach is known) is appropriate. The uncertainty associated with contaminant partitioning in Newtown Creek and the impacts of carbon normalization are discussed in Section 6.4.1. The impacts of carbon normalization on bioaccumulation are discussed in Section 6.6 and are described further in the *Modeling Approach Memorandum (3)* (MAM3; Anchor QEA 2018).

4.2.3 Distribution of Contaminants

The following subsections present information on the distribution of contaminants in surface sediment. Each of the contaminants evaluated for nature and extent is presented in a separate subsection, with the subsections organized by chemical class, with hydrocarbons (TPAH [17], TPAH [34], C19-C36) presented first, followed by bioaccumulative organics (TPCB and 2,3,7,8-TCDD), metals (Cu and Pb), and dieldrin (which is evaluated only for select media [i.e., surface sediment and tissue], as discussed previously). Discussions of nature and extent in other media (Sections 4.3 through 4.10) follow this same general organization.

4.2.3.1 TPAH (17)

TPAH (17) concentrations in surface sediment in each reach of the Study Area and the reference areas (including data from the 14 Phase 1 reference areas and the 4 Phase 2 reference areas) are summarized in Table 4-13. In addition, a plan view map, longitudinal profile, and box plot showing the data are provided in Figures 4-15 through 4-17, respectively. On the box plot (see Figure 4-17), Study Area surface sediment concentrations are compared with data from reference areas (including data collected during Phase 1 at the 14 Phase 1 reference areas and data collected during Phase 2 at the 4 Phase 2 reference areas).[89] Arithmetic average TPAH (17) concentrations within the Study Area reaches are generally consistent with those in the Phase 2 reference areas for the lower portions of the Study Area (i.e., CM 0 – 1, CM 1 – 2, and Whale Creek) and exceed the reference area concentrations in the upper reaches of the Study Area and tributaries.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. TPAH (17) concentrations range from 2.2 to 570 mg/kg (arithmetic average = 26 mg/kg; median = 15 mg/kg) and are generally the lowest in the Study Area (see Figure 4-17).
- **CM 1 – 2**. TPAH (17) concentrations range from 0.58 to 94 mg/kg (arithmetic average = 26 mg/kg; median = 25 mg/kg) and increase with distance upstream in CM 1 – 2.

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[89] The reference area surface sediment data are discussed in Section 4.1.1, with more detail in Sections 2.1.2 and 2.1.3.1.

- **CM 2+**. TPAH (17) concentrations range from 0.24 to 1,400 mg/kg (arithmetic average = 130 mg/kg; median = 72 mg/kg) and are generally the highest concentrations observed in the main stem.
- **English Kills**. TPAH (17) concentrations range from 5.9 to 290 mg/kg (arithmetic average = 77 mg/kg; median = 66 mg/kg). Concentrations in the lower 0.5 mile of English Kills are among the highest in the Study Area. From there, concentrations decline upstream to the head of the tributary.
- **East Branch**. TPAH (17) concentrations range from 2.4 to 370 mg/kg (arithmetic average = 61 mg/kg; median = 49 mg/kg) and are lower than those measured in the adjacent area of CM 2+.
- **Maspeth Creek**. TPAH (17) concentrations range from 5.6 to 180 mg/kg (arithmetic average = 62 mg/kg; median = 53 mg/kg) and are similar to the adjacent area of CM 2+.
- **Dutch Kills**. TPAH (17) concentrations range from 0.87 to 130 mg/kg (arithmetic average = 51 mg/kg; median = 54 mg/kg) and are higher than in the nearby main stem.
- **Whale Creek**. TPAH (17) concentrations range from 7.8 to 40 mg/kg (arithmetic average = 27 mg/kg; median = 30 mg/kg) and are slightly higher than in the nearby main stem.

4.2.3.2 TPAH (34)

TPAH (34) concentrations in surface sediment in each reach of the Study Area and the reference areas (based on data collected in the 14 Phase 1 reference areas and the 4 Phase 2 reference areas) are summarized in Table 4-14. In addition, a plan view map, longitudinal profile, and box plot showing the data are provided in Figures 4-18 through 4-20, respectively. Arithmetic average TPAH (34) concentrations within the Study Area reaches are generally higher than those in the Phase 2 reference areas.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. TPAH (34) concentrations range from 3.7 to 1,000 mg/kg (arithmetic average = 49 mg/kg; median = 27 mg/kg) and are generally the lowest in the Study Area (see Figure 4-20).

- **CM 1 – 2**. TPAH (34) concentrations range from 1.1 to 180 mg/kg (arithmetic average = 56 mg/kg; median = 46 mg/kg). While the concentrations in this reach are generally higher than those in several reference areas, there is overlap.
- **CM 2+**. TPAH (34) concentrations range from 0.55 to 5,400 mg/kg (arithmetic average = 340 mg/kg; median = 120 mg/kg). The highest concentrations in the main stem are observed in CM 2+.
- **English Kills**. TPAH (34) concentrations range from 9.6 to 1,900 mg/kg (arithmetic average = 240 mg/kg; median = 130 mg/kg). Concentrations in the lower 0.5 mile of English Kills are among the highest in the Study Area. From there, concentrations are generally lower upstream to the head of the tributary.
- **East Branch**. TPAH (34) concentrations range from 3.4 to 690 mg/kg (arithmetic average = 130 mg/kg; median = 86 mg/kg) and are generally lower than those measured in the adjacent area of CM 2+.
- **Maspeth Creek**. TPAH (34) concentrations range from 11 to 900 mg/kg (arithmetic average = 170 mg/kg; median = 110 mg/kg) and are similar to the adjacent area of CM 2+.
- **Dutch Kills**. TPAH (34) concentrations range from 2.2 to 350 mg/kg (arithmetic average = 100 mg/kg; median = 92 mg/kg) and are higher than in the nearby main stem.
- **Whale Creek**. TPAH (34) concentrations range from 19 to 240 mg/kg (arithmetic average = 49 mg/kg; median = 27 mg/kg) and are similar to or somewhat higher than in the nearby main stem.

4.2.3.3 C19-C36

C19-C36 concentrations in surface sediment in each reach of the Study Area and the reference areas (based on data collected in the four Phase 2 reference areas) are summarized in Table 4-15. In addition, a plan view map, longitudinal profile, and box plot showing the data are provided in Figures 4-21 through 4-23, respectively. C19-C36 was not analyzed in the Phase 1 or National Grid sampling programs; as such, there are fewer data available for this chemical in the Study Area and reference areas as compared to other chemicals. Nonetheless, the C19-C36 dataset is still large enough to support evaluations of nature and extent (it includes surface sediment samples from 240 Study Area locations and 40 reference area locations). Arithmetic average C19-C36 concentrations within the Study Area exceed Phase 2 reference area concentrations with the exception of CM 0 – 1 and CM 1 – 2.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. C19-C36 concentrations range from 16 to 250 mg/kg (arithmetic average = 65 mg/kg; median = 53 mg/kg). Surface sediment C19-C36 concentrations in CM 0 – 1 are generally the lowest in the Study Area (see Figure 4-23).
- **CM 1 – 2**. C19-C36 concentrations range from 23 to 1,000 mg/kg (arithmetic average = 210 mg/kg; median = 190 mg/kg). Concentrations in this reach are somewhat higher than those in CM 0 – 1 and increase in the upstream direction, especially at the upstream end of this reach (e.g., upstream of CM 1.8).
- **CM 2+**. C19-C36 concentrations range from 27 to 58,000 mg/kg (arithmetic average = 2,500 mg/kg; median = 920 mg/kg). A general increase in concentration from downstream to upstream is observed in this reach. The highest concentrations in the main stem are observed in CM 2+.
- **English Kills**. C19-C36 concentrations range from 7.7 to 31,000 mg/kg (arithmetic average = 2,900 mg/kg; median =1,500 mg/kg). Concentrations are similar to or higher than those measured in the adjacent area of CM 2+.
- **East Branch**. C19-C36 concentrations range from 35 to 7,300 mg/kg (arithmetic average = 2,000 mg/kg; median = 1,800 mg/kg). Concentrations are similar to those measured in the adjacent area of CM 2+.
- **Maspeth Creek**. C19-C36 concentrations range from 38 to 39,000 mg/kg (arithmetic average = 3,900 mg/kg; median = 1,200 mg/kg). Concentrations are similar to those measured in the adjacent area of CM 2+.
- **Dutch Kills**. C19-C36 concentrations range from 96 to 6,800 mg/kg (arithmetic average = 1,500 mg/kg; median = 1,300 mg/kg) and are substantially higher than those in the nearby main stem.
- **Whale Creek**. C19-C36 concentrations range from 91 to 960 mg/kg (arithmetic average = 500 mg/kg; median = 440 mg/kg) and are higher than in the nearby main stem.

4.2.3.4 TPCB

TPCB concentrations in surface sediment in each reach of the Study Area and the reference areas (including data from the 14 Phase 1 reference areas and the 4 Phase 2 reference areas) are summarized Table 4-16. In addition, a plan view map, longitudinal profile, and box plot showing the data are provided in Figures 4-24 through 4-26, respectively. Arithmetic average TPCB concentrations within the Study Area reaches are similar to those in all or some of the Phase 2 reference areas in CM 0 – 1 and CM 1 – 2, respectively, and generally exceed those for the reference areas elsewhere.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. TPCB concentrations range from 0.12 to 3.0 mg/kg (arithmetic average = 0.67 mg/kg; median = 0.62 mg/kg) and are generally the lowest in the Study Area (see Figure 4-26).
- **CM 1 – 2**. TPCB concentrations range from 0.098 to 3.0 mg/kg (arithmetic average = 1.1 mg/kg; median = 1.0 mg/kg) and are somewhat higher than in CM 0 – 1. While the concentrations in this reach are generally higher than those in several reference areas, there is overlap.
- **CM 2+**. TPCB concentrations range from 0.019 to 160 mg/kg (arithmetic average = 11 mg/kg; median = 6.5 mg/kg). An increase in concentration is observed above CM 2. The highest concentrations in the main stem are observed in CM 2+.
- **English Kills**. TPCB concentrations range from 0.11 to 63 mg/kg (arithmetic average = 12 mg/kg; median = 8.1 mg/kg). Concentrations in the lower 0.5 mile of English Kills are among the highest in the Study Area, but then decline upstream to the head of the tributary.
- **East Branch**. TPCB concentrations range from 0.024 to 16 mg/kg (arithmetic average = 2.4 mg/kg; median = 1.1 mg/kg). Concentrations are lower than in the adjacent area of CM 2+ and decline upstream to the head of the tributary.
- **Maspeth Creek**. TPCB concentrations range from 0.29 to 25 mg/kg (arithmetic average = 3.1 mg/kg; median = 1.4 mg/kg). Concentrations are lower than in the adjacent area of CM 2+ and do not exhibit a gradient, except for two higher values in the middle of the tributary.
- **Dutch Kills**. TPCB concentrations range from 0.015 to 380 mg/kg (arithmetic average = 15 mg/kg; median = 2.3 mg/kg). The majority of observed concentrations generally range from 1 to 10 mg/kg throughout the tributary, with the highest surface sediment TPCB concentration in the Study Area of 380 mg/kg located 0.2 mile upstream of the confluence with the main stem.

- **Whale Creek**. TPCB concentrations range from 0.36 to 3.4 mg/kg (arithmetic average = 1.4 mg/kg; median = 1.4 mg/kg). Concentrations are similar to the nearby main stem and do not exhibit a strong gradient, although concentrations tend to be higher closer to the head of the tributary.

4.2.3.5 2,3,7,8-TCDD

2,3,7,8-TCDD concentrations in surface sediment in each reach of the Study Area and the reference areas (including data from only the 4 Phase 2 reference areas) are summarized in Table 4-17. In addition, a plan view map, longitudinal profile, and box plot showing the data are provided in Figures 4-27 through 4-29, respectively. Arithmetic average 2,3,7,8-TCDD concentrations within the Study Area are generally consistent with those in the Phase 2 reference areas for some reaches (CM 0 – 1, East Branch, and Maspeth Creek), and generally exceed those for the reference areas elsewhere.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. 2,3,7,8-TCDD concentrations range from 0.85 to 11 nanograms per kilogram (ng/kg; arithmetic average = 5.4 ng/kg; median = 5.5 ng/kg) and are among the lowest in the Study Area.
- **CM 1 – 2**. 2,3,7,8-TCDD concentrations range from 0.33 to 23 ng/kg (arithmetic average = 7.8 ng/kg; median = 7.5 ng/kg) and are slightly higher than in CM 0 – 1.
- **CM 2+**. 2,3,7,8-TCDD concentrations range from 0.064 to 49 ng/kg (arithmetic average = 11 ng/kg; median = 9.0 ng/kg). An increase in concentration is observed above CM 2. The highest concentrations in the main stem are observed in CM 2+.
- **English Kills**. 2,3,7,8-TCDD concentrations range from 0.16 to 26 ng/kg (arithmetic average = 7.7 ng/kg; median = 6.0 ng/kg). Concentrations in the lower 0.5 mile of English Kills are among the highest in the Study Area, but then decline upstream to the head of the tributary.
- **East Branch**. 2,3,7,8-TCDD concentrations range from 0.12 to 14 ng/kg (arithmetic average = 3.2 ng/kg; median = 2.3 ng/kg). Concentrations near the confluence with the main stem are slightly higher than in the adjacent area of CM 2+ and decline upstream to the head of the tributary.

- **Maspeth Creek**. 2,3,7,8-TCDD concentrations range from 0.85 to 8.3 ng/kg (arithmetic average = 2.9 ng/kg; median = 2.7 ng/kg). Concentrations are lower than in the adjacent area of CM 2+.
- **Dutch Kills**. 2,3,7,8-TCDD concentrations range from 0.10 to 16 ng/kg (arithmetic average = 4.8 ng/kg; median = 3.6 ng/kg). Concentrations near the confluence with the main stem are slightly higher than in the adjacent area of the main stem, and then decline upstream to the head of the tributary.
- **Whale Creek**. 2,3,7,8-TCDD concentrations range from 4.8 to 65 ng/kg (arithmetic average = 19 ng/kg; median = 11 ng/kg). Concentrations near the confluence with the main stem are generally similar to the nearby main stem, but then increase upstream to the head of the tributary. Concentrations closer to the head of the tributary are among the highest in the Study Area and result in this tributary exhibiting the highest average concentration of any reach.

4.2.3.6 Cu

Cu concentrations in surface sediment in each reach of the Study Area and the reference areas (based on data collected in the 14 Phase 1 reference areas and the 4 Phase 2 reference areas) are summarized in Table 4-18. In addition, a plan view map, longitudinal profile, and box plot showing the data are provided in Figures 4-30 through 4-32, respectively. Arithmetic average Cu concentrations are similar to those in the Phase 2 reference areas for CM 0 – 1, and generally exceed those for the reference areas elsewhere.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. Cu concentrations range from 11 to 780 mg/kg (arithmetic average = 140 mg/kg; median = 120 mg/kg). Surface sediment Cu concentrations in CM 0 – 1 are generally the lowest in the Study Area (see Figure 4-32). Concentrations increase moving upstream with distance from the mouth.
- **CM 1 – 2**. Cu concentrations range from 60 to 650 mg/kg (arithmetic average = 290 mg/kg; median = 270 mg/kg). Concentrations continue to increase in the upstream direction.

- **CM 2+**. Cu concentrations range from 39 to 37,000 mg/kg (arithmetic average = 3,300 mg/kg; median = 2,200 mg/kg). An increase in concentration is observed above CM 2. The highest concentrations in the main stem are observed in CM 2+ at approximately CM 2.3.
- **English Kills**. Cu concentrations range from 60 to 4,300 mg/kg (arithmetic average = 860 mg/kg; median = 720 mg/kg). Concentrations are lower on average than concentrations measured in CM 2+, but are generally within the same range. Concentrations decline upstream to the head of the tributary.
- **East Branch**. Cu concentrations range from 32 to 6,300 mg/kg (arithmetic average = 570 mg/kg; median = 310 mg/kg). Concentrations are generally lower than the concentrations measured in CM 2+ and decline upstream to the head of the tributary.
- **Maspeth Creek**. Cu concentrations range from 72 to 5,900 mg/kg (arithmetic average = 730 mg/kg; median = 400 mg/kg). Concentrations are generally lower than in the adjacent area of CM 2+ and decline upstream to the head of the tributary.
- **Dutch Kills**. Cu concentrations range from 13 to 850 mg/kg (arithmetic average = 360 mg/kg; median = 350 mg/kg). Based on Figure 4-31b, concentrations are generally consistent throughout this tributary.
- **Whale Creek**. Cu concentrations range from 81 to 530 mg/kg (arithmetic average = 300 mg/kg; median = 280 mg/kg). Based on Figure 4-31b, concentrations are generally consistent throughout this tributary.

4.2.3.7 Pb

Pb concentrations in surface sediment in each reach of the Study Area and the reference areas (based on data collected in the 14 Phase 1 reference areas and the 4 Phase 2 reference areas) are summarized in Table 4-19. In addition, a plan view map, longitudinal profile, and box plot showing the data are provided in Figures 4-33 through 4-35, respectively. Arithmetic average Pb concentrations within the Study Area reaches are similar to those in the Phase 2 reference areas in CM 0 – 1 and CM 1 – 2, and generally exceed those for most reference areas elsewhere.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. Pb concentrations range from 10 to 750 mg/kg (arithmetic average = 130 mg/kg; median = 120 mg/kg). Surface sediment Pb concentrations in CM 0 – 1 are generally the lowest in the Study Area. Concentrations increase moving upstream with distance from the mouth.
- **CM 1 – 2**. Pb concentrations range from 34 to 3,100 mg/kg (arithmetic average = 270 mg/kg; median = 200 mg/kg). Concentrations continue to increase in the upstream direction.
- **CM 2+**. Pb concentrations range from 25 to 3,400 mg/kg (arithmetic average = 680 mg/kg; median = 600 mg/kg). An increase in concentration is observed above CM 2. The highest concentrations in the main stem are observed in CM 2+.
- **English Kills**. Pb concentrations range from 100 to 1,400 mg/kg (arithmetic average = 680 mg/kg; median = 640 mg/kg). Concentrations are similar to those measured in the adjacent area of CM 2+.
- **East Branch**. Pb concentrations range from 39 to 1,100 mg/kg (arithmetic average = 410 mg/kg; median = 380 mg/kg). Concentrations are similar to those measured in the adjacent area of CM 2+. Concentrations near the confluence with the main stem are generally consistent with the adjacent main stem.
- **Maspeth Creek**. Pb concentrations range from 56 to 1,300 mg/kg (arithmetic average = 370 mg/kg; median = 310 mg/kg). Concentrations are generally lower than in the adjacent area of CM 2+.
- **Dutch Kills**. Pb concentrations range from 9.0 to 1,200 mg/kg (arithmetic average = 440 mg/kg; median = 400 mg/kg). Concentrations generally decline slightly upstream to the head of the tributary.
- **Whale Creek**. Pb concentrations range from 76 to 430 mg/kg (arithmetic average = 240 mg/kg; median = 210 mg/kg). Concentrations are similar to (or greater than) the nearby main stem.

4.2.3.8 Dieldrin

Dieldrin concentrations in surface sediment in each reach of the Study Area and the reference areas (including data from the 14 Phase 1 reference areas and the 4 Phase 2 reference areas) are summarized in Table 4-20. In addition, a plan view map, longitudinal profile, and box plot showing the data are provided in Figures 4-36 through 4-38, respectively. Arithmetic average dieldrin concentrations are similar to those in the Phase 2 reference areas for CM 0 – 1 and CM 1 – 2, and generally exceed those for most reference areas elsewhere.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. Dieldrin concentrations range from 0.033 to 3.3 micrograms per kilogram (µg/kg; arithmetic average = 1.2 µg/kg; median = 1.3 µg/kg) and are generally the lowest in the Study Area.
- **CM 1 – 2**. Dieldrin concentrations range from 0.042 to 8.9 µg/kg (arithmetic average = 2.8 µg/kg; median = 2.3 µg/kg) and are slightly higher than in CM 0 – 1.
- **CM 2+**. Dieldrin concentrations range from 0.047 to 430 µg/kg (arithmetic average = 29 µg/kg; median = 16 µg/kg). An increase in concentration is observed above CM 2. The highest concentrations in the main stem are observed in CM 2+.
- **English Kills**. Dieldrin concentrations range from 0.12 to 200 µg/kg (arithmetic average = 45 µg/kg; median = 36 µg/kg). Concentrations in the lower 0.5 mile of English Kills are among the highest in the Study Area.
- **East Branch**. Dieldrin concentrations range from 0.066 to 130 µg/kg (arithmetic average = 20 µg/kg; median = 15 µg/kg). Concentrations near the confluence with the main stem are generally consistent with the adjacent main stem.
- **Maspeth Creek**. Dieldrin concentrations range from 0.030 to 84 µg/kg (arithmetic average = 17 µg/kg; median = 13 µg/kg). Concentrations near the confluence with the main stem are generally consistent with the adjacent main stem.
- **Dutch Kills**. Dieldrin concentrations range from 0.037 to 35 µg/kg (arithmetic average = 8.6 µg/kg; median = 5.8 µg/kg). Concentrations increase within the first 0.2 mile of the confluence with the main stem and then decline upstream to the head of the tributary.
- **Whale Creek**. Dieldrin concentrations range from 1.0 to 12 µg/kg (arithmetic average = 5.6 µg/kg; median = 6.4 µg/kg). Concentrations are higher or similar to those of the nearby main stem and are higher closer to the head of the tributary.

4.2.4 Impact of Recent NYC Navigational Dredging on Surface Sediment Chemical Concentrations

In 2014, NYC performed a navigational dredging project in the first mile of Newtown Creek and in Whale Creek, which included placement of a layer of sand cover following dredging.[90] Cores were collected during Phase 2 from within the 2014 NYC dredge program footprint (see Figure 4-39) to evaluate the potential impact of the 2014 NYC navigational dredging on concentrations of surface and subsurface sediment in the affected portion of the Study Area. Multiple samples were collected from 10 locations to characterize the sand cover material placed following dredging and the sediment layer just below the sand cover. Due to variability in the presence of sand cover material encountered during sampling, samples from the post-NYC dredge cores were separated into two general categories of observations for material at the surface, those with no discrete sand cover layer and those with variable sand cover layer (see Figure 4-39). Five locations had cores with varying amounts of sand cover material, ranging from 4 to 49 cm (approximately 2 to 20 inches) thick, as well as other cores with no observable sand cover layer (see Table 4-21). Three of these locations had cores with a sand cover layer below a layer of silt that ranged from 4 to 16 cm (approximately 2 to 6 inches) thick (see Table 4-21). The remaining five locations had no observable sand layer. Samples were generally taken from the top 0 to 15 cm (0 to 6 inches) for all cores at each location, regardless of the presence of sand cover material, to characterize the surface sediment. Subsurface sediment samples were taken only from cores with a sand cover layer present as a 15-cm (6-inch) sample interval that started immediately beneath the sand cover layer within each individual core.

Surface sediment concentrations from locations with no discrete sand cover are generally consistent with nearby Phase 1 and Phase 2 surface sediment data for the eight contaminants (or groups of contaminants) evaluated for nature and extent. Surface sediment concentrations at locations with variable sand cover are generally lower than nearby Phase 1 and Phase 2 surface sediment data for each of the eight contaminants or groups of contaminants (i.e., TPAH [17], TPAH [34], C19-C36, TPCB, 2,3,7,8-TCDD, Cu, Pb, and

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[90] The sand cover was placed approximately 1 to 2 weeks after dredging, with a target placement depth of at least 0.5 foot over all dredged areas (NYCDEP 2014a). NYC conducted a post-dredge, pre-sand placement bathymetry survey and a post-sand placement bathymetry survey to confirm that final target depths had been achieved after sand placement (NYCDEP 2014a).

dieldrin; see Tables 4-22a and 4-22b). Table 4-23a provides a listing of the types of samples collected at each location and the corresponding sediment concentrations for TPAH (17), TPAH (34), C19-C36, and TPCB, while concentrations for 2,3,7,8-TCDD, Cu, Pb, and dieldrin are provided in Table 4-23b. The concentrations of the lower layer of these variable sand cover cores (i.e., beneath the sand layer) are higher than the surface layer concentrations for each of the eight contaminants (or groups of contaminants) in nearly all cores (with the only exceptions being Cu and Pb in one core in the main stem and C19-C36 in one core in Whale Creek), which is anticipated from the placement of sand cover material over subsurface sediment that frequently has higher concentrations than the collocated surface sediment.

In general, the 2014 NYC navigational dredging does not appear to have resulted in significantly higher surface sediment concentrations within the dredge footprint, so it has not significantly impacted average concentrations of surface sediment in the reaches where the dredging occurred (CM 0 – 1 and Whale Creek).

4.2.5 PAH, PCB, and Metals Composition and Speciation

The preceding discussions of the nature and extent of PAHs, PCBs, and metals (Cu and Pb) were based on concentration results for the totals. Additional insight into nature and extent can be gained by evaluating the composition and speciation of these compounds within the surface sediment, as discussed in the subsections that follow.

4.2.5.1 PAHs

PAHs are a group of more than 1,000 individual compounds with two to six aromatic rings in various orientations, as well as alkyl substitutions. TPAH (17) and TPAH (34) are typical summations that are used to represent this class of compounds, as discussed in Section 4.1.3.2. Environmental samples may differ in composition (i.e., in the proportions of individual compounds), due to differences in source material and/or differences in weathering. Differences in composition are often evaluated by classifying PAHs into groups based on their structure. For example, the 17 PAHs in TPAH (17) (see Section 4.1.3.2) are split into low-molecular-weight polycyclic aromatic hydrocarbon (LPAH; 7 PAHs) and HPAH (10 PAHs) sums. TPAH (34) includes both the 17 compounds in TPAH (17), as well as 17 other C1- to C4-alkylated homologs of 2- to 6-ring PAHs.[91] The individual PAHs included in these sums are listed in Table 4-24.[92] The individual compounds included in these PAH groups have varying chemical properties, including solubility, partitioning strength (see Section 6.4.1), and toxicity.

TPAH (34) concentrations in Study Area surface sediment are higher than TPAH (17) concentrations, but the two exhibit very similar spatial patterns. This similarity is evident when comparing the two datasets directly using overlaid longitudinal profiles of individual sample results (see Figure 4-40), combined box plots by reach (see Figure 4-41), and a cross plot of individual sample results (Figure 4-42). These comparison figures show there is relatively good correlation and collocation between TPAH (17) and TPAH (34), which is to be expected because TPAH (17) is a component of TPAH (34). However, there is variability at the scale of individual samples, and the spatial gradients of the two longitudinal profiles differ slightly. This difference in spatial gradient is further illustrated by the longitudinal profile of the ratio of TPAH (34) to TPAH (17) in surface sediment presented in Figure 4-43.[93] Values of this ratio in CM 0 – 1 and in portions of the other main stem reaches and tributaries are within the range of values observed in the Phase 1 and Phase 2 reference area surface sediment, with a large fraction of samples in the upper portions of the Study area and tributaries exhibiting higher values for this ratio. Reaches where the ratio of TPAH (34) to TPAH (17) is relatively higher generally coincide with the reaches where concentrations of TPAH (34) are relatively higher, indicating a greater fraction of alkylated PAH compounds in those higher concentration areas. Longitudinal profiles of HPAH and LPAH concentrations are also generally similar to TPAH (17) and TPAH (34) (comparing Figures 4-44 and 4-45 with Figures 4-16a, 4-16b, 4-19a, and 4-19b), which again is to be expected given the degree of overlap in compounds included in these different sums. The values for the ratio LPAH/HPAH (see Figure 4-46) are mostly within the range of reference

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[91] Samples collected as part of the National Grid (2009/2010) sampling were not analyzed for TPAH (34).

[92] Longitudinal profiles, box plots, and summary tables of concentrations for each measured individual PAH are available in Attachment A-A of Appendix A.

[93] Samples that were collected as part of the triad program are not included in Figures 4-40 through 4-42 and 4-43 through 4-46. Triad samples were analyzed for PAHs by a different laboratory (Hawthorne) than the rest of the Phase 1 and Phase 2 data (Alpha). Data from both laboratories satisfied quality control requirements. Nonetheless, the concentrations of some PAH sums differ between these two datasets, primarily due to a difference in the laboratory methodology. Similar insights can be drawn from these two datasets. The triad program data are included in TPAH (17) analyses presented throughout this report.

area data, with values that exceed reference area data found in some samples from CM 2+ and a few tributaries.

4.2.5.2 PCBs

PCBs are a group of 209 individual chlorinated compounds or congeners consisting of a biphenyl molecule and one to ten chlorine atoms. They are grouped according to the number of chlorine atoms per biphenyl molecule (CBP), which ranges from one (monochlorobiphenyl) to ten (decachlorobiphenyl). These groups are termed homologs. PCB congeners vary in their chemical properties, which affect both their fate and transport and bioaccumulation characteristics. For example, PCBs with more chlorine atoms tend to sorb more strongly to particulate matter.

PCBs were produced as specific mixtures of congeners; in the United States, the most common were produced under the tradename "Aroclor." Common examples are Aroclor 1242 (12 carbon atoms; 42 wt% chlorine) and Aroclor 1260 (12 carbon atoms; 60 wt% chlorine). PCBs from multiple sources are commonly found in the environment. Thus, multiple Aroclors are often identified in environmental samples.

In addition, PCBs that have been released over time weather in the environment through various mechanisms, including physical-chemical processes (partitioning, in conjunction with dissolved phase and particulate phase transport processes) and biologically based dechlorination. This means that the composition of the PCBs in an environmental sample can be a complex mixture of congeners. Thus, it may be difficult to identify the original Aroclors that contributed to an environmental sample.[94]

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[94] In addition, identification of Aroclors, even for the same sample, can vary based on chromatogram interpretation. Quantification of Aroclors is achieved by identifying a small group of peaks in the chromatogram that correspond to each Aroclor. Some peaks are shared between Aroclors, and identification relies on peak ratios and pattern recognition. Furthermore, as Aroclors weather in the environment, the peak patterns change, making individual Aroclors less recognizable. Aroclor identification in these instances often relies on the analyst's experience and expertise, and the approach to the identification can vary from laboratory to laboratory and analyst to analyst. Differences among analysts in the interpretation of the chromatogram, combined with changes in composition that may occur due to weathering, can lead to variable Aroclor identification for the same sample. For these reasons, PCB composition is evaluated in this report using only the congener data.

To provide a general understanding of the variability in PCB composition in the Study Area, the composition of PCBs in the surface sediment of the Study Area and the four Phase 2 reference areas is evaluated using a simple metric: the CBP from each sample. This metric provides only a crude assessment of PCB composition; a single number cannot be used to fully explore differences among samples with more than 100 congeners. For example, two samples with differing Aroclor composition may exhibit similar values for CBP. Furthermore, weathering can change the value of CBP over time at a given location. Nonetheless, this metric provides a tool to explore gross differences in PCB composition throughout the Study Area. More extensive evaluation will be provided during development of the CFT, which will consider individual PCB homologs (see the *Modeling Approach Memorandum (2)* [MAM2; Anchor QEA 2016c]).

Figure 4-47 presents a longitudinal profile of CBP in surface sediment, along with the four Phase 2 reference areas (surface sediment congener data were not collected in Phase 1 reference areas). The horizontal dashed lines in this figure show the values of CBP for pure samples of the most common Aroclors; note that only qualitative comparisons can be made with the Aroclors, due to the potential weathering in environmental samples. The first observation from this figure is that there are differences in composition throughout the Study Area. Elevated CBP values are observed in the upstream portion of Dutch Kills, Maspeth Creek, and a few samples from other reaches. The surface sediment sample with the highest TPCB concentration in the Study Area, 380 mg/kg (sample DK049), was collected in Dutch Kills and exhibits elevated CBP, along with other samples in its vicinity. CBP in Dutch Kills and Maspeth Creek is generally similar and generally exceeds values measured in the reference areas. This pattern suggests that PCB sources to these tributaries included more heavily chlorinated congeners than sources to the rest of the Study Area. Most samples from the main stem, Whale Creek, East Branch, and English Kills exhibit CBP that generally lies within the range of reference area data. CBP is somewhat higher in East Branch than in English Kills. In the main stem, CBP increases gradually upstream from the mouth, peaks in the area of CM 2.0 and 2.6, and then declines toward the head of English Kills. This gradient may be due to weathering or to mixing of differing source Aroclors. In summary, the PCB congener mixture in Study Area surface sediment varies spatially and is suggestive of more than one Aroclor source, as well as past and potentially ongoing mixing and/or weathering.

Analysis of the Phase 1 Aroclor data provides a similar picture (see Attachment A-A1 of Appendix A). Most of the PCBs in the Study Area were classified as a mixture of Aroclors 1242 and 1254, and the highest concentrations were found in CM 2+ and English Kills, similar to TPCB congeners. Elevated concentrations of Aroclor 1260 were observed in Dutch Kills; this is consistent with the elevated TPCB congener concentrations, as well as the observation of higher CBP values, in Dutch Kills. Thus, the Aroclor data support the conclusion (based on CBP) that PCB composition varies throughout the Study Area, likely indicative of different source materials.

4.2.5.3 Simultaneously Extracted Metals

The composition of metals in the Study Area surface sediment was evaluated. For the divalent metals Cu, Cd, Pb, Ni, and Zn, bulk sediment AVS and SEM are often used to predict toxicity to benthic macroinvertebrates (Di Toro et al. 1992; Ankley et al. 1996; Berry et al. 1996). The AVS present in sediment reacts with these metals, forming insoluble metal sulfides and thereby reducing bioavailability. In the Phase 2 field programs, bulk sediment AVS and SEM were analyzed in surface sediment grab samples to support the evaluation of metal bioavailability and toxicity in the BERA. These data were collected for the sediment quality triad (SQT) samples (see Section 7.2) in the Study Area and the Phase 2 reference areas, as well as additional sediment locations sampled for benthic community analysis in the Study Area and the Phase 2 reference areas. The bioavailability of SEM was evaluated using the sum of SEM minus AVS (Σ SEM – AVS).

As shown in Figure 4-48, Σ SEM – AVS is less than zero for all Study Area triad sample locations, indicating reduced bioavailability for SEM and a lack of metals bioavailability with respect to bulk sediment exposures.[95] Note that inclusion of TOC as a binding phase would further decrease the estimated metals bioavailability throughout the Study Area.

The stability of AVS and SEM in bulk sediment was evaluated during the toxicity tests by evaluating Σ SEM – AVS in the in situ, pre-test, and post-test samples. Because there was no

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[95] Bioavailability of SEM using AVS is typically evaluated using the following relationship: (Σ SEM – AVS)/f_{OC} (USEPA 2005b). Because Σ SEM – AVS is less than zero for all Study Area triad sample locations, including the f_{OC} term in the calculation will also result in values less than 0 micromoles per gram (µmol/gram) OC; this is also less than the no-effect value of 130 µmol/gram OC as reported in USEPA (2005b).

statistically significant change between pre-test and post-test results for Σ SEM – AVS, AVS and SEM were stable during the course of the toxicity tests. This indicates that the manipulations of the sediments that occurred during toxicity testing and the introduction of oxygen to the sediments that would have resulted from these manipulations did not impact AVS measurement significantly.[96] Although in situ Σ SEM – AVS is statistically significantly different (more negative) than both pre-test and post-test Σ SEM – AVS, the slight increase in the bioavailable fraction during the toxicity tests was still well below chronic threshold values.

To further refine the evaluation of metals bioavailability, a subset of the Phase 2 triad sediment samples was selected for metals speciation. Following initial mineral identification using X-ray diffraction, a further subset of the samples was selected for sequential extraction (Tessier et al. 1979) and for electron microprobe analysis (Reed 2005). Sequential extraction is designed to fractionate metals in a sample according to their reactivity/bioavailability, by subjecting the sample to a sequence of progressively more aggressive chemical treatments that target specific chemical forms. Electron microprobe analysis allows determination of grain-scale mineralogy and trace metal distribution to support interpretation of the sequential extraction data.

Metals speciation also supports the conclusion of a lack of metals bioavailability with respect to bulk sediment exposures (see Attachment E2 of the BERA). Very few metals were found to be present in an exchangeable form in the sediment samples that were evaluated. Exchangeable metals would be assumed to be bioavailable. However, even when exchangeable metals were found in a sample, the percentages that were exchangeable were low relative to the percentage of the metal that was insoluble in the same sample. Therefore, the concentrations of bioavailable forms of metals are very low in the Study Area surface sediment samples.

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[96] The introduction of DO to sediments can result in the oxidation of sulfide and a reduction in AVS. Thus, had the introduction of oxygen been significant, oxygen levels in the sediments would have increased to a significant extent, AVS would have decreased, and metals bioavailability would have increased during the process of setting up and conducting the toxicity tests. The finding that AVS exceeded SEM throughout the testing process indicates that any introduction of any oxygen to the sediments during processing had a *de minimis* effect on metals bioavailability.

4.3 Subsurface Sediment

4.3.1 Subsurface Sediment Dataset

The RI subsurface sediment dataset includes samples that were collected during the Phase 1, Phase 2, and National Grid sampling, and the shoreline sediment sampling efforts of Part 1 of the FS field program (see Table 2-2b). Most subsurface samples were collected as cores that extended through the subsurface sediment into the top of the native material. Depending on the objective of the sampling program, subsurface samples were either collected continuously throughout the core, or collected at selected intervals sampled within the core. Most of the cores were collected to evaluate broadscale horizontal and vertical patterns within the Study Area, so they were segmented continuously (i.e., the entire core was segmented and analyzed), with segments approximately 1 to 3 feet long. One Phase 2 program, the high-resolution core sampling program, is discussed in this section separately from the rest of the subsurface data, because these high-resolution cores were segmented every 2 cm (approximately 1 inch) within the upper 2 feet of sediment. The subsurface core results are discussed in the subsections that follow. Typical core lengths extend from 60-cm depth in the sediment bed to approximately 600 cm (2 to 20 feet).

4.3.2 Percent Fines and Total Organic Carbon

4.3.2.1 Percent Fines

The distribution of the fine sediment content (i.e., percent fines) in Study Area subsurface sediment is presented in Table 4-25 and Figures 4-49[97] and 4-50. Across the Study Area, the arithmetic average percent fines in subsurface sediment is 56 wt%, although individual samples range from near zero to near 100 wt% and arithmetic average values by reach and depth interval range from 39 wt% to 77 wt% (excluding reaches and depths with fewer than 10 samples).[98] Reach-by-reach median values are presented in Table 4-25 and Figure 4-50 and range from 43 wt% to 90 wt% across the Study Area reaches and depth intervals having greater than 10 samples. Lower fine sediment contents are generally found at the upstream ends of the

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[97] Figure 4-49 and subsequent subsurface sediment figures contain panels showing native material as well. These are discussed in Section 4.4.

[98] Reaches and depths with fewer than 10 samples may not accurately reflect the central tendency of the concentrations given the limited sample count and, therefore, are excluded from the range discussed in this sentence. Arithmetic averages that are outside of the range of 39% to 77% are from reaches and depths with only two samples.

tributaries near the CSO discharges (except for Whale Creek). This is consistent with the facts that CSO effluent solids contain a mixture of coarse and fine material, and coarse material tends to deposit closer to its source than fine material. Vertically, the sediment core profile maps in Figures 4-51a through 4-51j[99] show variability in fine sediment content, with values generally being higher in the shallower portions and decreasing with depth in many cores.

4.3.2.2 Total Organic Carbon

The distribution of the TOC in Study Area subsurface sediment is presented in Table 4-26 and Figures 4-52 and 4-53. Individual sample values generally range from 5 wt% to 25 wt%, with some above 30 wt% (see Table 4-26 and Figures 4-52 and 4-53) found in CM 2+. Across the Study Area, the arithmetic average TOC in subsurface sediment is 11.4 wt%, with arithmetic average values by reach and depth interval ranging from 5.9 wt% to 17 wt% (excluding reaches and depths with fewer than 10 samples).[100] Reach-by-reach median values for subsurface TOC range from 4.4 wt% to 21 wt% across the Study Area reaches and depth intervals having greater than 10 samples; these values appear to be generally high, compared with the surface sediment data from the Phase 1 and Phase 2 reference areas (see Figure 4-53). TOC concentrations within the subsurface sediment increase with depth in some reaches, generally by a factor of approximately 2 over depths of 2 to 4 meters (see Figure 4-54), with variability in the vertical pattern among individual cores (see Figures 4-55a through 4-55j). This pattern suggests higher historical organic loads from industrial and municipal sources combined with the depositional nature of the system.[101]

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[99] In this figure and others like it, the bar plots show the results measured in the paired surface sediment grab and all subsurface sediment depth intervals of the corresponding core using a color scale (divided into quintiles of the data). The location at the top of the bar plot corresponds to the geographical coordinates of the surface sediment grab sample, with all the subsurface sediment core depth intervals stacked beneath.

[100] Reaches and depths with fewer than 10 samples may not accurately reflect the central tendency of the concentrations given the limited sample count and, therefore, are excluded from the range discussed in this sentence. Arithmetic averages that are outside of the range of 5.9% to 17% are from reaches and depths with only four or fewer samples.

[101] Industrial facilities that may have historically contributed organic materials to the sediments of Newtown Creek include adhesives, animal rendering, asphalt production, coal processing, creosote production, distilleries, incinerators, manufactured gas plants, metal production, metal scrap, paints and pigments, paper products, petroleum refining, plastics, printing, railyards, sawmills, shipbuilding, solid waste disposal, utilities, and waste oil refining (see Section 3.2.6 for a more detailed discussion).

4.3.2.3 Total Organic Carbon Composition

As discussed in Section 4.2.2.4, TPH is an analytical parameter that includes a complex mixture of thousands of individual petroleum hydrocarbon compounds, from a variety of sources, and constitutes a portion of TOC. TPH concentrations in subsurface sediment generally range from 10^3 to 10^5 mg/kg (0.1 wt% to 10 wt% of dry sediment). Concentrations increase moving upstream from the mouth in CM 0 – 1. TPH concentrations decline toward the upstream ends of English Kills, Maspeth Creek, and Dutch Kills (see Figure 4-56). The arithmetic average of the ratio of TPH to TOC is 0.25 in subsurface sediment, and the ratio generally ranges from 0.05 to 0.50 (see Figure 4-57). TPH concentrations within the subsurface sediment generally increase with depth, up to a factor of approximately 10 (see Figure 4-58), although there is variability among cores (see Figures 4-59a through 4-59j).

Soot carbon concentrations range from approximately 0.1 wt% to 2 wt% of dry matter in subsurface sediment (see Figure 4-60). In general, soot carbon is spread fairly evenly throughout the Study Area, with a few high values scattered along the main stem and in CM 2+. The arithmetic average of the ratio of soot carbon to TOC is 0.12 in subsurface sediment; data range generally from 0.01 to 0.20, with some higher values (see Figure 4-61[102]). Soot carbon concentrations within the subsurface sediment increase somewhat with depth, up to a factor of approximately 2 (see Figures 4-62 through 4-63j), indicating that historically, there were higher inputs of soot carbon to the Study Area.

As with surface sediment, the composition of subsurface sediment TOC is important for CFT, in particular for determining the partitioning behavior of chemicals. Partitioning and its impacts on CFT in subsurface sediment are discussed in Section 6.4.

4.3.3 Distribution of Contaminants

4.3.3.1 TPAH (17)

TPAH (17) concentrations in subsurface sediment in each reach of the Study Area are summarized in Table 4-27. A box plot, longitudinal profile, box plot depth profile, and stacked bar maps showing the data are provided in Figures 4-64 through 4-67j,

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[102] A small number of samples have a ratio of soot carbon to TOC greater than 1, due to uncertainty in concentrations near the MDL.

respectively.[103] Throughout the Study Area, TPAH (17) concentrations in subsurface sediment are generally higher than concentrations in surface sediment (comparing Figures 4-65 to Figures 4-16a and 4-16b) and generally exhibit similar longitudinal patterns. In general, subsurface sediment TPAH (17) concentrations increase with depth below the surface in all regions of the Study Area, although there is variability among individual core locations (see Figures 4-66 through 4-67j and Table 4-27; note that these figures include surface sediment data and subsurface sediment data from cores that were sampled continuously and do not include samples within native material).

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. TPAH (17) concentrations range from 0.59 mg/kg (60 to 100 cm) to 11,000 mg/kg (200 to 300 cm), arithmetic averages range from 34 mg/kg (400 to 500 cm) to 1,300 mg/kg (200 to 300 cm), and medians range from 22 mg/kg (15 to 60 cm) to 140 mg/kg (200 to 300 cm). Concentrations in CM 0 – 1 are generally the lowest near the mouth of the Study Area and increase upstream. Arithmetic average TPAH (17) concentrations are generally similar to upstream reaches, except for CM 2+.
- **CM 1 – 2**. TPAH (17) concentrations range from 26 mg/kg (15 to 60 cm) to 2,800 mg/kg (100 to 200 cm), arithmetic averages range from 85 mg/kg (300 to 400 cm) to 880 mg/kg (200 to 300 cm), and medians range from 70 mg/kg (15 to 60 cm) to 690 mg/kg (200 to 300 cm). Concentrations are generally similar to CM 0 – 1.
- **CM 2+**. TPAH (17) concentrations range from 0.11 mg/kg (60 to 100 cm) to 49,000 mg/kg (100 to 200 cm), and arithmetic averages range from 450 mg/kg (60 to 100 cm) to 7,100 mg/kg (300 to 400 cm), and medians range from 170 mg/kg (15 to 60 cm) to 1000 mg/kg (300 to 400 cm). Within the main stem, the highest concentrations are in CM 2+, between CM 2.3 and 2.7.
- **English Kills**. TPAH (17) concentrations range from 1.8 mg/kg (15 to 60 cm) to 1,900 mg/kg (100 to 200 cm), arithmetic averages range from 220 mg/kg (60 to 100 cm) to 330 mg/kg (100 to 200 cm), and medians range from 140 mg/kg (60 to 100 cm) to 310

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[103] The sample counts in the table and figures may differ because of differences in types of subsurface sediment cores. The convention used for this RI Report is that the tables are based on the data from continuous cores only, whereas some figures (such as the box plot) include additional cores from which not all depth intervals were analyzed.

mg/kg (400 to 500 cm). Subsurface sediment concentrations at the mouth are similar to nearby main stem concentrations and decrease upstream toward the head of the tributary.

- **East Branch**. TPAH (17) concentrations range from 12 mg/kg (300 to 400 cm) to 3,100 mg/kg (100 to 200 cm), arithmetic averages range from 160 mg/kg (15 to 60 cm) to 430 mg/kg (100 to 200 cm), and medians range from 110 mg/kg (15 to 60 cm) to 230 mg/kg (200 to 300 cm). Subsurface sediment concentrations at the mouth are similar to nearby main stem concentrations and decrease upstream toward the head of the tributary.
- **Maspeth Creek**. TPAH (17) concentrations range from 13 mg/kg (15 to 60 cm) to 1,700 mg/kg (400 to 500 cm), arithmetic averages range from 160 mg/kg (60 to 100 cm) to 910 mg/kg (300 to 400 cm), and medians range from 110 mg/kg (15 to 60 cm) to 860 mg/kg (300 to 400 cm). Concentrations are similar to nearby main stem concentrations.
- **Dutch Kills**. TPAH (17) concentrations range from 8.8 mg/kg (400 to 500 cm) to 2,100 mg/kg (300 to 400 cm), arithmetic averages range from 100 mg/kg (15 to 60 cm) to 580 mg/kg (300 to 400 cm), and medians range from 53 mg/kg (400 to 500 cm) to 260 mg/kg (300 to 400 cm). Subsurface sediment concentrations at the mouth are similar to nearby main stem concentrations and decrease upstream toward the head of the tributary.
- **Whale Creek**. TPAH (17) concentrations range from 91 mg/kg (15 to 60 cm) to 1,300 mg/kg (100 to 200 cm), arithmetic averages range from 92 mg/kg (15 to 60 cm) to 830 mg/kg (100 to 200 cm), and medians range from 92 mg/kg (15 to 60 cm) to 830 mg/kg (100 to 200 cm). Concentrations are similar to nearby main stem concentrations.

4.3.3.2 TPAH (34)

TPAH (34) concentrations in subsurface sediment in each reach of the Study Area are summarized in Table 4-28. A box plot, longitudinal profile, box plot depth profile, and stacked bar maps showing the data are provided in Figures 4-68 through 4-71j, respectively. Throughout the Study Area, TPAH (34) concentrations in subsurface sediment are generally higher than concentrations in surface sediment (comparing Figures 4-69 to 4-19a and 4-19b) and generally exhibit similar longitudinal patterns. In general, subsurface sediment TPAH (34) concentrations increase with depth below the surface in all regions of the Study Area, although there is variability among the individual core locations (see Figures 4-70 through 4-71j and Table 4-28; note that these figures include surface sediment data and subsurface sediment data from cores that were sampled continuously and do not include samples within native material).

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. TPAH (34) concentrations range from 1.1 mg/kg (60 to 100 cm) to 14,000 mg/kg (200 to 300 cm), arithmetic averages range from 86 mg/kg (400 to 500 cm) to 1,800 mg/kg (200 to 300 cm), and medians range from 39 mg/kg (15 to 60 cm) to 310 mg/kg (200 to 300 cm). Median concentrations in CM 0 – 1 are generally the lowest in the Study Area.
- **CM 1 – 2**. TPAH (34) concentrations range from 47 mg/kg (15 to 60 cm) to 4,400 mg/kg (100 to 200 cm), arithmetic averages range from 220 mg/kg (300 to 400 cm) to 2,000 mg/kg (200 to 300 cm), and medians range from 150 mg/kg (15 to 60 cm) to 2,100 mg/kg (200 to 300 cm). Concentrations are generally similar to CM 0 – 1.
- **CM 2+**. TPAH (34) concentrations range from 0.24 mg/kg (60 to 100 cm) to 15,000 mg/kg (100 to 200 cm), arithmetic averages range from 1,100 mg/kg (15 to 60 cm) to 4,500 mg/kg (500 to 600 cm), and medians range from 850 mg/kg (15 to 60 cm) to 2,400 mg/kg (300 to 400 cm). Some of the highest TPAH (34) concentrations in the main stem are in CM 2+
- **English Kills**. TPAH (34) concentrations range from 3.6 mg/kg (300 to 400 cm) to 3,200 mg/kg (60 to 100 cm and 100 to 200 cm), arithmetic averages range from 620 mg/kg (15 to 60 cm) to 1,200 mg/kg (400 to 500 cm), and medians range from 440 mg/kg (15 to 60 cm and 60 to 100 cm) to 1,200 mg/kg (400 to 500 cm). Subsurface sediment concentrations at the mouth are similar to nearby main stem concentrations and decrease upstream toward the head of the tributary.
- **East Branch.** TPAH (34) concentrations range from 21 mg/kg (15 to 60 cm) to 6,100 mg/kg (100 to 200 cm), arithmetic averages range from 440 mg/kg (15 to 60 cm) to 1,100 mg/kg (100 to 200 cm), and medians range from 280 mg/kg (15 to 60 cm) to 740 mg/kg (200 to 300 cm). Subsurface sediment concentrations at the mouth are similar to nearby main stem concentrations and decrease upstream toward the head of the tributary.

- **Maspeth Creek**. TPAH (34) concentrations range from 25 mg/kg (15 to 60 cm) to 3,900 mg/kg (400 to 500 cm), arithmetic averages range from 550 mg/kg (60 to 100 cm) to 2,100 mg/kg (300 to 400 cm), and medians range from 370 mg/kg (15 to 60 cm) to 2,300 mg/kg (300 to 400 cm). Concentrations are similar to nearby main stem concentrations near the mouth and show some decrease toward the head of the tributary.
- **Dutch Kills**. TPAH (34) concentrations range from 23 mg/kg (400 to 500 cm) to 3,400 mg/kg (200 to 300 cm), arithmetic averages range from 330 mg/kg (15 to 60 cm) to 1,000 mg/kg (300 to 400 cm), and medians range from 140 mg/kg (400 to 500 cm) to 650 mg/kg (100 to 200 cm). Subsurface sediment concentrations at the mouth are similar to nearby main stem concentrations and decrease upstream toward the head of the tributary.
- **Whale Creek**. TPAH (34) concentrations range from 240 mg/kg (15 to 60 cm) to 2,200 mg/kg (100 to 200 cm), and arithmetic averages and medians both range from 240 mg/kg (15 to 60 cm) to 1,800 mg/kg (100 to 200 cm), based on data from a limited number of continuous cores (two). Concentrations are similar to nearby main stem concentrations.

4.3.3.3 C19-C36

C19-C36 concentrations in subsurface sediment in each reach of the Study Area are summarized in Table 4-29. A box plot, longitudinal profile, box plot depth profile, and stacked bar maps showing the data are provided in Figures 4-72 through 4-75j, respectively. C19-C36 was not analyzed in Phase 1 or National Grid sampling programs; thus, there are fewer cores that were sampled continuously and analyzed for C19-C36 as compared to other chemicals (e.g., there are no cores that were sampled continuously and analyzed for C19-C36 in CM 1 – 2, Maspeth Creek, or Whale Creek). As such, although there are more than 100 subsurface sediment samples analyzed for C19-C36 to evaluate general concentration ranges, the number of continuous cores with which to assess vertical patterns is limited in several reaches. In reaches where there are available continuous cores to evaluate vertical patterns, C19-C36 concentrations in subsurface sediment are generally higher than concentrations in surface sediment. In general, subsurface sediment C19-C36 concentrations increase with depth below the surface throughout the Study Area (other than in some tributaries), although there is variability among individual core locations (see Figures 4-74 through 4-75j and Table 4-29).

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. C19-C36 concentrations range from 150 mg/kg (15 to 60 cm) to 2,700 mg/kg (100 to 200 cm), and arithmetic averages and medians both range from 420 mg/kg (15 to 60 cm) to 2,100 mg/kg (100 to 200 cm), based on data from a limited number of continuous cores (two). Concentrations in CM 0 – 1 are generally the lowest in the Study Area and show evidence of an increase from CM 0 on the downstream end to CM 1 on the upstream end.
- **CM 1 – 2**. No cores were sampled continuously for C19-C36 in CM 1 – 2 to allow for detailed evaluation of vertical patterns. Concentrations from the discontinuous core sections that were analyzed for C19-C36 are generally similar to those in CM 0 – 1.
- **CM 2+**. C19-C36 concentrations range from 150 mg/kg (200 to 300 cm) to 9,600 mg/kg (60 to 100 cm), arithmetic averages range from 1,900 mg/kg (400 to 500 cm) to 5,500 mg/kg (60 to 100 cm), and medians range from 2,000 mg/kg (400 to 500 cm) to 6,200 mg/kg (60 to 100 cm). Within the main stem, the highest concentrations are in CM 2+.
- **English Kills**. C19-C36 concentrations range from 11 mg/kg (300 to 400 cm) to 16,000 mg/kg (15 to 60 cm), arithmetic averages range from 4,700 mg/kg (200 to 300 cm) to 7,900 mg/kg (300 to 400 cm), and medians range from 3,500 mg/kg (200 to 300 cm) to 12,000 mg/kg (300 to 400 cm). Concentrations in English Kills are generally the highest in the Study Area.
- **East Branch.** C19-C36 concentrations range from 370 mg/kg (60 to 100 cm) to 7,600 mg/kg (400 to 500 cm), arithmetic averages range from 700 mg/kg (60 to 100 cm) to 4,900 mg/kg (400 to 500 cm), and medians range from 700 mg/kg (15 to 60 cm and 60 to 100 cm) to 4,200 mg/kg (400 to 500 cm). Subsurface sediment concentrations have a range that is similar to those in the nearby main stem, whereas the median and arithmetic average are lower.
- **Maspeth Creek**. There are no subsurface sediment data for C19-C36 in Maspeth Creek.
- **Dutch Kills**. C19-C36 concentrations range from 280 mg/kg (15 to 60 cm) to 2,000 mg/kg (200 to 300 cm), arithmetic averages range from 400 mg/kg (15 to 60 cm) to 1,500 mg/kg (200 to 300 cm), and medians range from 400 mg/kg (15 to 60 cm) to 1,400 mg/kg (200 to 300 cm). Subsurface sediment concentrations are similar to the nearby main stem.

- **Whale Creek**. There is only one subsurface sample for C19-C36 in Whale Creek, with results being non-detect.

4.3.3.4 TPCB

TPCB concentrations in subsurface sediment in each reach of the Study Area are summarized in Table 4-30. A box plot, longitudinal profile, box plot depth profile, and stacked bar maps showing the data are provided in Figures 4-76 through 4-79j, respectively. Throughout the Study Area, TPCB concentrations in subsurface sediment are generally higher than concentrations in surface sediment (comparing Figures 4-77 to Figures 4-25a and 4-25b). Subsurface sediment TPCB concentrations, when considering all depths, vary by approximately three orders of magnitude and exhibit longitudinal spatial patterns that are similar to those in surface sediment.

In general, subsurface sediment TPCB concentrations increase with depth below the surface in all regions of the Study Area, although there is variability among individual core locations (see Figures 4-78 through 4-79j and Table 4-30; note that these figures include surface sediment data and subsurface sediment data from cores that were sampled continuously and do not include samples within native material). In CM 1 – 2, CM 2+, and Dutch Kills, concentrations peak within the top 2 meters of the subsurface sediment bed and decline toward the bottom, suggesting that the deepest portions of the subsurface sediment were deposited prior to more substantive historical PCB releases.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. TPCB concentrations range from 0.001 mg/kg (60 to 100 cm) to 8.1 mg/kg (100 to 200 cm), arithmetic averages range from 0.47 mg/kg (300 to 400 cm) to 2.5 mg/kg (100 to 200 cm), and medians range from 0.44 mg/kg (300 to 400 cm) to 1.5 mg/kg (100 to 200 cm). Concentrations in CM 0 – 1 are generally the lowest in the Study Area.
- **CM 1 – 2**. TPCB concentrations range from 8.8E-5 mg/kg (300 to 400 cm) to 30 mg/kg (15 to 60 cm), arithmetic averages range from 1.3 mg/kg (200 to 300 cm) to

6.6 mg/kg (60 to 100 cm), and medians range from 0.30 mg/kg (200 to 300 cm) to 4.2 mg/kg (60 to 100 cm). Concentrations are higher than in CM 0 – 1.

- **CM 2+**. TPCB concentrations range from 7.3E-6 mg/kg (60 to 100 cm) to 190 mg/kg (60 to 100 cm), arithmetic averages range from 1.3 mg/kg (400 to 500 cm) to 33 mg/kg (60 to 100 cm), and medians range from 0.20 mg/kg (400 to 500 cm) to 19 mg/kg (60 to 100 cm). Some of the highest TPCB concentrations in the main stem are in CM 2+, between CM 2.5 and 2.7.
- **English Kills**. TPCB concentrations range from 0.0032 mg/kg (200 to 300 cm) to 170 mg/kg (200 to 300 cm), arithmetic averages range from 29 mg/kg (15 to 60 cm) to 70 mg/kg (400 to 500 cm), and medians range from 24 mg/kg (15 to 60 cm) to 70 mg/kg (400 to 500 cm). Some of the highest subsurface sediment TPCB concentrations are in English Kills. Similar to surface sediment, English Kills TPCB concentrations are highest near CM 3.0 to 3.3 and decrease upstream to the head of the tributary.
- **East Branch**. TPCB concentrations range from 0.056 mg/kg (200 to 300 cm) to 46 mg/kg (100 to 200 cm), arithmetic averages range from 7.7 mg/kg (300 to 400 cm) to 13 mg/kg (100 to 200 cm and 400 to 500 cm), and medians range from 5.4 mg/kg (200 to 300 cm) to 12 mg/kg (400 to 500 cm). Concentrations are generally lower than the nearby main stem.
- **Maspeth Creek**. TPCB concentrations range from 0.003 mg/kg (60 to 100 cm) to 37 mg/kg (200 to 300 cm), arithmetic averages range from 2.2 mg/kg (400 to 500 cm) to 12 mg/kg (200 to 300 cm), and medians range from 1.4 mg/kg (400 to 500 cm) to 10 mg/kg (60 to 100 cm). Concentrations are lowest near the head of the tributary.
- **Dutch Kills**. TPCB concentrations range from 0.0032 mg/kg (400 to 500 cm) to 35 mg/kg (60 to 100 cm and 100 to 200 cm), arithmetic averages range from 0.35 mg/kg (400 to 500 cm) to 14 mg/kg (60 to 100 cm), and medians range from 0.023 mg/kg (400 to 500 cm) to 11 mg/kg (60 to 100 cm). The highest subsurface sediment TPCB concentrations are in the middle portion of the tributary and decrease downstream and upstream to the head of the tributary, consistent with the surface sediment.
- **Whale Creek**. TPCB concentrations range from 0.012 mg/kg (200 to 300 cm) to 11 mg/kg (60 to 100 cm), arithmetic averages range from 0.9 mg/kg (200 to 300 cm) to 6 mg/kg (60 to 100 cm), and medians range from 0.90 mg/kg (200 to 300 cm) to 6.0 mg/kg (60 to 100 cm). Concentrations are similar to the nearby main stem.

4.3.3.5 2,3,7,8-TCDD

2,3,7,8-TCDD concentrations in subsurface sediment in each reach of the Study Area are summarized in Table 4-31. A box plot, longitudinal profile, box plot depth profile, and stacked bar maps showing the data are provided in Figures 4-80 through 4-83j, respectively. Throughout the Study Area, 2,3,7,8-TCDD concentrations in subsurface sediment are generally higher than concentrations in surface sediment (comparing Figure 4-81 to Figures 4-28a and 4-28b) and generally exhibit similar longitudinal patterns. In general, subsurface sediment 2,3,7,8-TCDD concentrations increase with depth below the surface in all regions of the Study Area, although there is variability among individual core locations (see Figures 4-82 through 4-83j and Table 4-31; note that these figures include surface sediment data and subsurface sediment data from cores that were sampled continuously and do not include samples within native material).

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. 2,3,7,8-TCDD concentrations range from 0.071 ng/kg (60 to 100 cm) to 68 ng/kg (100 to 200 cm), arithmetic averages range from 7.8 ng/kg (15 to 60 cm) to 24 ng/kg (100 to 200 cm), and medians range from 4.0 ng/kg (200 to 300 cm) to 18 ng/kg (100 to 200 cm). Concentrations in CM 0 – 1 are generally the lowest in the Study Area, and within the reach, decrease somewhat with distance upstream.
- **CM 1 – 2**. 2,3,7,8-TCDD concentrations range from 0.35 ng/kg (300 to 400 cm) to 46 ng/kg (100 to 200 cm), arithmetic averages range from 2.9 ng/kg (200 to 300 cm) to 21 ng/kg (60 to 100 cm), and medians range from 2.9 ng/kg (200 to 300 cm) to 18 ng/kg (15 to 60 cm). Concentrations are higher than in CM 0 – 1 and are generally consistent within this reach.
- **CM 2+**. 2,3,7,8-TCDD concentrations range from 0.083 ng/kg (60 to 100 cm) to 150 ng/kg (100 to 200 cm), arithmetic averages range from 3.6 ng/kg (400 to 500 cm) to 51 ng/kg (100 to 200 cm), and medians range from 1.9 ng/kg (400 to 500 cm) to 45 ng/kg (100 to 200 cm). The highest 2,3,7,8-TCDD concentrations in the main stem are in CM 2+, from approximately CM 2.3 to 2.7.
- **English Kills**. 2,3,7,8-TCDD concentrations range from 0.25 ng/kg (200 to 300 cm) to 60 ng/kg (100 to 200 cm), arithmetic averages range from 17 ng/kg (15 to 60 cm) to

34 ng/kg (400 to 500 cm), and medians range from 16 ng/kg (15 to 60 cm) to 34 ng/kg (400 to 500 cm). Concentrations are similar to the nearby main stem and decrease somewhat with distance upstream to the head of the tributary.

- **East Branch.** 2,3,7,8-TCDD concentrations range from 0.34 ng/kg (15 to 60 cm) to 28 ng/kg (400 to 500 cm), arithmetic averages range from 2.2 ng/kg (100 to 200 cm) to 15 ng/kg (400 to 500 cm), and medians range from 1.4 ng/kg (100 to 200 cm) to 11 ng/kg (400 to 500 cm). Concentrations are similar to the nearby main stem and are generally consistent within this reach.
- **Maspeth Creek**. 2,3,7,8-TCDD concentrations range from 0.31 ng/kg (100 to 200 cm) to 50 ng/kg (200 to 300 cm), arithmetic averages range from 6.8 ng/kg (400 to 500 cm) to 22 ng/kg (200 to 300 cm), and medians range from 6.8 ng/kg (400 to 500 cm) to 14 ng/kg (60 to 100, 200 to 300, and 300 to 400 cm). Concentrations generally decrease upstream to the head of the tributary.
- **Dutch Kills**. 2,3,7,8-TCDD concentrations range from 1.1 ng/kg (60 to 100 cm) to 18 ng/kg (15 to 60 cm), arithmetic averages range from 5.5 ng/kg (100 to 200 cm) to 7.6 ng/kg (60 to 100 cm), and medians range from 2.8 ng/kg (15 to 60 cm) to 7.2 ng/kg (60 to 100 cm). Concentrations are generally similar throughout this reach (recognizing the limited number of sample locations).
- **Whale Creek**. No continuous cores collected from this reach were analyzed for 2,3,7,8-TCDD.

4.3.3.6 Cu

Cu concentrations in subsurface sediment in each reach of the Study Area are summarized in Table 4-32. A box plot, longitudinal profile, box plot depth profile, and stacked bar maps showing the data are provided in Figures 4-84 through 4-87j, respectively. Throughout the Study Area, Cu concentrations in subsurface sediment are generally higher than concentrations in surface sediment (comparing Figure 4-85 to Figures 4-31a and 4-31b). Subsurface sediment Cu concentrations, when considering all depths, vary by approximately two orders of magnitude and exhibit longitudinal spatial patterns that are similar to those in surface sediment. In general, subsurface sediment Cu concentrations increase with depth below the surface in all regions of the Study Area, although there is variability among individual core locations (see Figures 4-86 through 4-87j and Table 4-32); note that these figures include surface sediment data and subsurface sediment data from cores that were sampled continuously and do not include samples within native material).

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. Cu concentrations range from 9.3 mg/kg (60 to 100 cm) to 1,300 mg/kg (60 to 100 cm and 100 to 200 cm), arithmetic averages range from 280 mg/kg (15 to 60 cm) to 600 mg/kg (100 to 200 cm), and medians range from 230 mg/kg (15 to 60 cm) to 550 mg/kg (100 to 200 cm). Concentrations in CM 0 – 1 are generally the lowest in the Study Area.
- **CM 1 – 2**. Cu concentrations range from 37 mg/kg (100 to 200 cm) to 6,000 mg/kg (60 to 100 cm), arithmetic averages range from 150 mg/kg (300 to 400 cm) to 2,300 mg/kg (100 to 200 cm), and medians range from 150 mg/kg (300 to 400 cm) to 1,900 mg/kg (100 to 200 cm). Concentrations are higher than in CM 0 – 1 and increase with distance upstream.
- **CM 2+**. Cu concentrations range from 20 mg/kg (60 to 100 cm) to 170,000 mg/kg (15 to 60 cm), arithmetic averages range from 4,400 mg/kg (500 to 600 cm) to 10,000 mg/kg (60 to 100 cm), and medians range from 2,300 mg/kg (500 to 600 cm) to 8,700 mg/kg (100 to 200 cm). The highest Cu concentrations in the main stem are in CM 2+, near CM 2.3.
- **English Kills**. Cu concentrations range from 12 mg/kg (300 to 400 cm) to 7,600 mg/kg (100 to 200 cm), arithmetic averages range from 2,100 mg/kg (15 to 60 cm) to 4,900 mg/kg (400 to 500 cm), and medians range from 1,800 mg/kg (15 to 60 cm) to 4,900 mg/kg (400 to 500 cm). Concentrations are similar to the nearby main stem and East Branch.
- **East Branch.** Cu concentrations range from 180 mg/kg (60 to 100 cm) to 6,000 mg/kg (100 to 200 cm), arithmetic averages range from 1,600 mg/kg (15 to 60 cm) to 2,500 mg/kg (100 to 200 cm), and medians range from 1,200 mg/kg (15 to 60 cm) to 2,300 mg/kg (60 to 100 cm). Concentrations generally decrease upstream to the head of the tributary.
- **Maspeth Creek**. Cu concentrations range from 420 mg/kg (60 to 100 cm) to 13,000 mg/kg (400 to 500 cm), arithmetic averages range from 3,400 mg/kg (15 to 60 cm) to 7,900 mg/kg (300 to 400 cm), and medians range from 3,200 mg/kg (15 to 60

cm) to 9,800 mg/kg (200 to 300 cm). Concentrations generally decrease upstream to the head of the tributary.

- **Dutch Kills**. Cu concentrations range from 46 mg/kg (400 to 500 cm) to 4,200 mg/kg (200 to 300 cm), arithmetic averages range from 530 mg/kg (400 to 500 cm) to 1,600 mg/kg (100 to 200 cm), and medians range from 470 mg/kg (400 to 500 cm) to 1,900 mg/kg (60 to 100 cm). The highest subsurface sediment Cu concentrations are in the middle portion of the tributary and decrease downstream and upstream to the head of the tributary, consistent with the surface sediment.
- **Whale Creek**. Cu concentrations range from 200 mg/kg (200 to 300 cm) to 2,700 mg/kg (60 to 100 cm), and arithmetic averages range from 910 mg/kg (15 to 60 cm) to 2,100 mg/kg (60 to 100 cm), and medians range from 910 mg/kg (15 to 60 cm) to 2,100 mg/kg (60 to 100 cm). Concentrations are similar to the nearby main stem.

4.3.3.7 Pb

Pb concentrations in subsurface sediment in each reach of the Study Area are summarized in Table 4-33. A box plot, longitudinal profile, box plot depth profile, and stacked bar maps showing the data are provided in Figures 4-88 through 4-91j, respectively. Throughout the Study Area, Pb concentrations in subsurface sediment are generally higher than concentrations in surface sediment (comparing Figure 4-89 to Figures 4-34a and 4-34b). Subsurface sediment Pb concentrations, when considering all depths, vary by approximately two orders of magnitude and generally exhibit longitudinal spatial patterns that are similar to those in surface sediment (i.e., lowest concentrations in CM 0 – 1, with increases upstream within the main stem and the upstream tributaries). In general, subsurface sediment Pb concentrations increase with depth below the surface to some extent in all regions of the Study Area, although there is variability among individual core locations (see Figures 4-90 through 4-91j and Table 4-33); note that these figures include surface sediment data and subsurface sediment data from cores that were sampled continuously and do not include samples within native material).

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. Pb concentrations range from 3.8 mg/kg (60 to 100 cm) to 1,300 mg/kg (200 to 300 cm), arithmetic averages range from 220 mg/kg (15 to 60 cm) to 470 mg/kg (200 to 300 cm and 400 to 500 cm), and medians range from 190 mg/kg (15 to 60 cm) to 470 mg/kg (400 to 500 cm). Concentrations in CM 0 – 1 are generally the lowest in the Study Area.
- **CM 1 – 2**. Pb concentrations range from 28 mg/kg (100 to 200 cm) to 3,200 mg/kg (200 to 300 cm), arithmetic averages range from 110 mg/kg (300 to 400 cm) to 1,200 mg/kg (200 to 300 cm), and medians range from 110 mg/kg (300 to 400 cm) to 870 mg/kg (100 to 200 cm). Concentrations are higher than in CM 0 – 1 and increase with distance upstream.
- **CM 2+**. Pb concentrations range from 6.5 mg/kg (60 to 100 cm) to 2,300 mg/kg (300 to 400 cm), arithmetic averages range from 770 mg/kg (300 to 400 cm) to 1,300 mg/kg (60 to 100 cm and 100 to 200 cm), and medians range from 690 mg/kg (300 to 400 cm) to 1,400 mg/kg (60 to 100 cm). The highest Pb concentrations in the main stem are in CM 2+.
- **English Kills**. Pb concentrations range from 5.3 mg/kg (300 to 400 cm) to 4,000 mg/kg (60 to 100 cm), arithmetic averages range from 1,200 mg/kg (15 to 60 cm) to 1,900 mg/kg (400 to 500 cm), and medians range from 1,200 mg/kg (15 to 60 cm) to 1,900 mg/kg (300 to 400 cm and 400 to 500 cm). Concentrations are similar to the nearby main stem and East Branch.
- **East Branch.** Pb concentrations range from 79 mg/kg (60 to 100 cm) to 2,400 mg/kg (100 to 200 cm), arithmetic averages range from 920 mg/kg (15 to 60 cm) to 1,400 mg/kg (400 to 500 cm), and medians range from 810 mg/kg (15 to 60 cm) to 1,500 mg/kg (100 to 200 cm). Concentrations are similar to the nearby main stem and English Kills.
- **Maspeth Creek**. Pb concentrations range from 180 mg/kg (15 to 60 cm) to 2,200 mg/kg (200 to 300 cm), arithmetic averages range from 1,000 mg/kg (15 to 60 cm) to 1,600 mg/kg (200 to 300 cm), and medians range from 1,200 mg/kg (15 to 60 cm) to 1,900 mg/kg (200 to 300 cm). Concentrations generally decrease upstream to the head of the tributary.
- **Dutch Kills**. Pb concentrations range from 40 mg/kg (400 to 500 cm) to 2,200 mg/kg (60 to 100 cm and 200 to 300 cm), arithmetic averages range from 330 mg/kg (400 to 500 cm) to 1,200 mg/kg (60 to 100 cm), and medians range from 310 mg/kg (400 to

500 cm) to 1,200 mg/kg (60 to 100 cm). Concentrations generally decrease upstream to the head of the tributary.

- **Whale Creek**. Pb concentrations range from 190 mg/kg (200 to 300 cm) to 1,700 mg/kg (60 to 100 cm), arithmetic averages and medians range from 520 mg/kg (200 to 300 cm) to 1,200 mg/kg (60 to 100 cm). Concentrations are similar to the nearby main stem.

4.3.4 Near-Surface Vertical Patterns

Contaminant concentrations within the surface sediment (the top 15 cm [6 inches] of the sediment bed) reflect a combination of recently deposited contaminated solids that have been mixed into historically deposited legacy contamination (see Section 6.4.4). Understanding deposition and mixing is an important focus of the RI and provides information for effective decision-making (see Sections 6 and 8). High-resolution cores were collected during Phase 2 to provide a snapshot of vertical contaminant patterns within the surface sediment (top 0- to 15-cm [0- to 6-inch] increment) and shallow subsurface (the 15- to 60-cm [6- to 24-inch] increment of the sediment bed) at locations throughout the Study Area (see Figures 4-92a through 4-92d). In addition, the more coarsely sectioned cores (see Figures 4-93 through 4-99) permit a comparison of the surface sediment with the shallow subsurface sediment (the more coarsely sectioned cores generally contained a segment extending from 15 to 60 cm [6 to 24 inches]), which similarly provides insights into deposition and mixing. Coarser-scaled vertical patterns are described first, followed by the finer-scaled patterns observed in the high-resolution cores.

4.3.4.1 Coarsely Sectioned Deep Continuous Cores

Cross plots of surface sediment concentrations (0 to 15 cm [0 to 6 inches]) versus concentrations in the first subsurface sediment sample from each sediment core (usually the 15- to 60-cm [6- to 24-inch] depth interval) are presented in Figures 4-93 through 4-99 for TPAH (17), TPAH (34), C19-C36, TPCB, Cu, 2,3,7,8-TCDD, and Pb, respectively. In nearly every core, the concentration of the surface sediment sample is lower than the first subsurface sediment sample. For TPAH (17), TPAH (34), and TPCB, this pattern is consistent throughout the Study Area, with the exception of CM 0 – 1, where surface and subsurface sediment concentrations are generally similar, and the concentrations are generally lower. For Cu and

Pb, surface sediment concentrations are lower than the corresponding first subsurface sediment samples in all but a few cores throughout the entire Study Area (see Figures 4-98 and 4-99). For C19-C36 and 2,3,7,8-TCDD, surface sediment concentrations are lower than the corresponding first subsurface sediment interval samples in most Study Area sample locations, other than some relatively low concentration samples from Dutch Kills and English Kills (see Figures 4-95 and 4-97). The observation that surface sediment concentrations are generally lower than subsurface sediment concentrations is confirmed by a p value of < 0.05 for the binomial statistical test in all seven chemicals (see Figures 4-93 through 4-99). This result indicates that solids with lower contaminant concentrations (compared to those in the existing sediment bed) have been depositing on the sediment bed during more recent years. This concentration difference has led to an overall decrease in surface sediment concentrations throughout the Study Area (see Section 6.4.4.5 for further discussion).

4.3.4.2 High-Resolution Cores

High-resolution cores were collected throughout the Study Area, including the main stem and all five tributaries. In Figures 4-92a through 4-92d, each row presents the data for a single chemical, and each column presents the data for a single core. Cores are arranged from downstream to upstream in the Study Area (core ID and CM are provided at the top of each column of plot panels). Each core was segmented in 2-cm (approximately 1-inch) intervals down to a depth of 60 cm (2 feet); the following 2-cm intervals were analyzed in the laboratory: 0 to 2, 2 to 4, 4 to 6, 8 to 10, 18 to 20, 28 to 30, and 58 to 60 cm.

Throughout the Study Area, the concentration of the first sample interval (0 to 2 cm [0 to 1 inch]) is consistently lower than the deepest sample (58 to 60 cm [2 feet]) for each of the five chemicals evaluated (i.e., TPAH [17], TPAH [34], TPCB, Cu, and Pb; 2,3,7,8-TCDD and C19-C36 were not included in the analyte list for these cores). This pattern corroborates the results of the coarsely sectioned cores described above, namely that over the years represented by these segments, depositing solids have had lower contaminant concentrations than previously deposited surface sediment.

Notable patterns within the high-resolution cores are as follows:

- **CM 0 – 1** (cores NC154, NC161, and NC259). TPAH (17), TPAH (34), TPCB, Cu, and Pb concentrations in these cores do not exhibit consistent vertical patterns in the top 30 cm (12 inches). Some cores exhibit increases just below the surface (i.e., between the 0 to 2 and 2 to 4 cm segments) for some chemicals (followed by decreases in some cases), but others do not, recognizing that the concentration range is relatively low for some of the chemicals. Regardless, in all cases, the highest concentrations in each core are present at the approximately 60-cm (24-inch) interval. Contaminant concentrations are low in this reach and within the range of the reference area data (see Section 4.2.3).
- **CM 1 – 2** (cores NC037 and NC169). The two cores located in CM 1 – 2 exhibit consistent increases with depth for each of the five contaminants (with the exception of TPAH [17] and TPAH [34] in the topmost segment in core NC037, and TPCB, Cu, and Pb in the bottommost segment in core NC169). The vertical concentration gradient in the upper 20 cm of core NC169 differs somewhat between TPAH (17) and TPAH (34), with the gradient being smoother for TPAH (34); however, both chemicals exhibit the same overall increase with depth.
- **CM 2+** (NC174 and NC071), **English Kills** (EK006), **East Branch** (EB006), and **Maspeth Creek** (MC005). Core NC174 (CM 2.35) exhibits consistent increases with depth (with the exception of the deepest TPCB value), similar to the cores from CM 1 – 2. In Core NC071, all five chemicals exhibit a similar pattern, with relatively low to moderate concentrations in the top 10 cm, a single elevated concentration at 20 cm, and lower concentrations below 20 cm. Core MC005 does not exhibit a consistent pattern with depth. The cores from East Branch (EB006) and English Kills (EK006) exhibit generally increasing concentration trends with depth, with the highest concentrations generally at the deepest depth for all five chemicals (with the exception of TPAH [17] and TPAH [34] in Core EK006).
- **Dutch Kills** (DK037). In the one core in Dutch Kills, TPCB, TPAH (17), TPAH (34), Cu, and Pb concentrations generally increase with depth, with some variability in a few segments for some of the chemicals (e.g., TPAH [17], Cu, and Pb). TPAH (17), TPAH (34), and TPCB concentrations in this core are elevated (in contrast to nearby CM 0 – 1, where surface sediment contains similar concentrations to those of the reference areas).

- **Whale Creek** (WC012). The one core in Whale Creek shows a relatively flat concentration profile in the top 10 cm (4 inches) for all five contaminants, with concentrations generally increasing below that (with some variability). The highest concentrations of TPAH (34), TPCB, Cu, and Pb are in the deepest segment.

Additional evaluation of the high-resolution cores with respect to deposition and mixing processes is provided in Section 6.4.4.

4.4 Native Material

4.4.1 Native Material Dataset

The native material RI dataset includes samples that were collected during the Phase 1, Phase 2, and National Grid sampling events (see Table 2-2b). Native material samples were collected in cores that extended through the subsurface sediment into the top of the native material, based on observed lithology (see Section 3.1.2).

4.4.2 Percent Fines, Total Organic Carbon, TPH, and Soot Carbon

4.4.2.1 Percent Fines

With respect to percent fines, native material differs substantially from subsurface sediment. Percent fines range from less than 1 wt% to 100 wt% (see Figures 4-49 and 4-50), with no consistent spatial pattern in the Study Area, although values are generally lower than in subsurface sediment (e.g., for reaches with more than three native material samples, median percent fine values are much lower for native material than for subsurface sediment in all reaches, except CM 0 – 1).

4.4.2.2 Total Organic Carbon

Similar to percent fines, TOC concentrations within native material differs substantially from those in subsurface sediment. TOC concentrations are low in native material (ranging from 0.016 wt% to 20 wt%, with 94% of the values less than 2 wt%; see Figures 4-52 and 4-53) as compared to subsurface sediment. Of 209 native material samples, only 5 have TOC exceeding 4 wt% (1 in CM 0 – 1, 1 in Dutch Kills, and 3 in the Turning Basin).

4.4.2.3 Total Organic Carbon Composition

TPH concentrations range from 0.56 to 54,000 mg/kg (see Figure 4-56). Eighty-five percent of the data are less than 100 mg/kg, which is generally one to three orders of magnitude lower than subsurface sediment data. Eleven of the seventeen values exceeding 100 mg/kg are found in CM 1 – 2 and in English Kills. TPH exceeding 10,000 mg/kg was measured in two native material samples in lower English Kills.[104] The ratio of TPH to TOC in native material is lower than that in subsurface sediment. The arithmetic average of the ratio of TPH to TOC is 0.051 in native material, and the ratio generally ranges from 0.001 to 0.1, with some higher and some lower values (see Figure 4-57).

Similar to TOC, soot carbon concentrations in native material are low (ranging from 0.01 wt% to 8.3 wt%; see Figure 4-60). Ninety-three percent of the values in native material are less than 1.0 wt%. Somewhat elevated values (as high as 3 wt%, except for one value at 8 wt%) are found in the Turning Basin. The ratio of soot carbon to TOC in native material is higher than that in subsurface sediment. The arithmetic average of the ratio of soot carbon to TOC is 0.64 in native material; data range generally from 0.1 to 2.0, with some higher values (see Figure 4-61).[105]

4.4.3 Distribution of Contaminants

4.4.3.1 TPAH (17)

TPAH (17) concentrations in native material in each reach of the Study Area are shown in the box plot and longitudinal profile provided in Figures 4-64 and 4-65, respectively. Concentrations in native material are generally two to three orders of magnitude lower than subsurface sediment (see Figure 4-64).

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

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[104] Although elevated TPH concentrations may sometimes be considered a potential indication of NAPL, the presence of NAPL was evaluated using other LOEs and approaches in the RI, as detailed in Section 4.6.1 (see also Section 5 of Appendix C).

[105] Some samples have a ratio of soot carbon to TOC greater than 1, due to uncertainties in concentrations near the MDL.

- **CM 0 – 1**. TPAH (17) concentrations range from 0.0088 to 74 mg/kg (arithmetic average = 5.1 mg/kg; median = 0.27 mg/kg).
- **CM 1 – 2**. TPAH (17) concentrations range from 0.00063 to 38 mg/kg (arithmetic average = 3.0 mg/kg; median = 0.20 mg/kg) and are the lowest of any main stem segment.
- **CM 2+**. TPAH (17) concentrations range from 0.0051 to 5,400 mg/kg (arithmetic average = 350 mg/kg; median = 0.74 mg/kg). A limited number of samples exceeding 10 mg/kg are found in CM 2+, where higher subsurface sediment concentrations are also found. These native material concentrations lie within the range of subsurface sediment concentrations in the same region.
- **English Kills**. TPAH (17) concentrations range from 0.0011 to 15,000 mg/kg (arithmetic average = 530 mg/kg; median = 0.13 mg/kg). A limited number of samples exceeding 10 mg/kg are found in lower English Kills, where higher subsurface sediment concentrations are also found. These native material concentrations lie within the range of subsurface sediment concentrations in the same region. Concentrations decline upstream to the head of English Kills.
- **East Branch**. TPAH (17) concentrations range from 0.0011 to 39 mg/kg (arithmetic average = 3.9 mg/kg; median = 0.20 mg/kg).
- **Maspeth Creek**. TPAH (17) concentrations range from 0.036 to 0.27 mg/kg (arithmetic average = 0.15 mg/kg; median = 0.14 mg/kg).
- **Dutch Kills**. TPAH (17) concentrations range from 0.0078 to 170 mg/kg (arithmetic average = 19 mg/kg; median = 0.94 mg/kg).
- **Whale Creek**. TPAH (17) concentrations range from 0.0056 to 9.2 mg/kg (arithmetic average = 2.2 mg/kg; median = 0.83 mg/kg).

Additional discussion of the distribution of contaminants in subsurface sediment and native material, in particular relating to the delineation and distribution of potential NAPL, is provided in Section 4.6.1 (see also Sections 3.3 through 3.5 and Section 5 of Appendix C).

4.4.3.2 TPAH (34)

TPAH (34) concentrations in native material in each reach of the Study Area are shown in the box plot and longitudinal profile provided in Figures 4-68 and 4-69, respectively. Concentrations in native material are generally two to three orders of magnitude lower than subsurface sediment (see Figure 4-68).

The range, arithmetic average, and median for each reach are as follows (no notable patterns were identified in any reach, except for English Kills):

- **CM 0 – 1**. TPAH (34) concentrations range from 0.0075 to 88 mg/kg (arithmetic average = 6.8 mg/kg; median = 0.45 mg/kg).
- **CM 1 – 2**. TPAH (34) concentrations range from 0.0020 to 150 mg/kg (arithmetic average = 9.9 mg/kg; median = 0.44 mg/kg).
- **CM 2+**. TPAH (34) concentrations range from 0.0091 to 110 mg/kg (arithmetic average = 5.4 mg/kg; median = 0.28 mg/kg).
- **English Kills**. TPAH (34) concentrations range from 0.0031 to 22,000 mg/kg (arithmetic average = 780 mg/kg; median = 0.31 mg/kg). A limited number of samples exceeding 10 mg/kg are found in lower English Kills, where higher subsurface sediment concentrations are also found. These native material concentrations lie within the range of subsurface sediment concentrations in the same region. Concentrations decline upstream to the head of English Kills.
- **East Branch**. TPAH (34) concentrations range from 0.0060 to 92 mg/kg (arithmetic average = 9.5 mg/kg; median = 0.44 mg/kg).
- **Maspeth Creek**. TPAH (34) concentrations range from 0.12 to 0.53 mg/kg (arithmetic average = 0.35 mg/kg; median = 0.37 mg/kg).
- **Dutch Kills**. TPAH (34) concentrations range from 0.023 to 480 mg/kg (arithmetic average = 53 mg/kg; median = 1.7 mg/kg).
- **Whale Creek**. TPAH (34) concentrations range from 0.012 to 40 mg/kg (arithmetic average = 8.6 mg/kg; median = 1.6 mg/kg).

4.4.3.3 C19-C36

C19-C36 concentrations in native material in each reach of the Study Area are shown in the box plot and longitudinal profile provided in Figures 4-72 and 4-73, respectively. Concentrations in native material are frequently non-detect (FoD of 21%), with detected values being generally one or two orders of magnitude lower than subsurface sediment (see Figure 4-72), other than a limited number of samples with concentrations greater than 100 mg/kg in some of the tributaries (see Figure 4-73).

The range, arithmetic average, and median for each reach are as follows (no notable patterns were identified in any reach):

- **CM 0 – 1**. C19-C36 concentrations range from 3.8 to 65 mg/kg (arithmetic average = 19 mg/kg; median = 15 mg/kg).
- **CM 1 – 2**. C19-C36 concentrations range from 7.5 to 50 mg/kg (arithmetic average = 13 mg/kg; median = 8.6 mg/kg).
- **CM 2+**. C19-C36 was detected in one sample from this tributary, at a concentration of 71 mg/kg.
- **English Kills**. C19-C36 concentrations range from 7.1 to 1,000 mg/kg (arithmetic average = 57 mg/kg; median = 7.9 mg/kg).
- **East Branch**. C19-C36 concentrations range from 7.3 to 290 mg/kg (arithmetic average = 47 mg/kg; median = 8.4 mg/kg).
- **Maspeth Creek**. C19-C36 was not detected in any of the samples from this tributary.
- **Dutch Kills**. C19-C36 concentrations range from 8.0 to 1,700 mg/kg (arithmetic average = 500 mg/kg; median = 8.8 mg/kg).
- **Whale Creek**. C19-C36 concentrations range from 7.3 to 610 mg/kg (arithmetic average = 330 mg/kg; median = 380 mg/kg).

4.4.3.4 TPCB

TPCB concentrations in native material in each reach of the Study Area are shown in the box plot and longitudinal profile provided in Figures 4-76 and 4-77, respectively. Sixty-six percent of the TPCB samples in native material were non-detect. Furthermore, TPCB concentrations in native material are generally two to three orders of magnitude lower than surface sediment within most of the Study Area and are generally lower than concentrations in the reference area surface sediment (see Figure 4-77 and comparing Figures 4-25a and 4-25b).

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. TPCB concentrations range from 2.6E-5 to 0.94 mg/kg (arithmetic average = 0.043 mg/kg; median = 0.0010 mg/kg).

- **CM 1 – 2**. TPCB concentrations range from 4.2E-6 to 0.26 mg/kg (arithmetic average = 0.0096 mg/kg; median = 0.0018 mg/kg) and are the lowest of any main stem segment.
- **CM 2+**. TPCB concentrations range from 2.3E-5 to 130 mg/kg (arithmetic average = 2.2 mg/kg; median = 0.021 mg/kg). A limited number of samples exceeding 1 mg/kg are found in CM 2+, between CM 2.5 and 2.7, where higher subsurface sediment concentrations are also found. These native material concentrations lie within the range of subsurface sediment concentrations in the same region.
- **English Kills**. TPCB concentrations range from 3.9E-5 to 0.79 mg/kg (arithmetic average = 0.045 mg/kg; median = 0.0039 mg/kg).
- **East Branch**. TPCB concentrations range from 5.0E-5 to 0.27 mg/kg (arithmetic average = 0.028 mg/kg; median = 0.0036 mg/kg).
- **Maspeth Creek**. TPCB was not detected in any of the samples from this tributary.
- **Dutch Kills**. TPCB concentrations range from 9.9E-5 to 6.9 mg/kg (arithmetic average = 0.64 mg/kg; median = 0.0048 mg/kg).
- **Whale Creek**. TPCB was detected in one sample from this tributary, at a concentration of 0.077 mg/kg.

4.4.3.5 2,3,7,8-TCDD

2,3,7,8-TCDD concentrations in native material in each reach of the Study Area are shown in the box plot and longitudinal profile provided in Figures 4-80 and 4-81, respectively. FoD for 2,3,7,8-TCDD in native material samples was only 5%. Furthermore, 2,3,7,8-TCDD concentrations in native material are generally one to three orders of magnitude lower than surface sediment concentrations within most of the Study Area, and are generally lower than concentrations in the reference area surface sediment (see Figure 4-81 and comparing Figures 4-29a and 4-29b).

Detected concentrations of 2,3,7,8-TCDD in native material (if any) for each reach are as follows:

- **CM 0 – 1**. 2,3,7,8-TCDD was detected in one sample, at a concentration of 7.7 ng/kg.
- **CM 1 – 2**. 2,3,7,8-TCDD was detected in one sample, at a concentration of 2.8 ng/kg.
- **CM 2+**. 2,3,7,8-TCDD was detected in one sample, at a concentration of 1.9 ng/kg.

- **English Kills, East Branch, and Maspeth Creek.** 2,3,7,8-TCDD was not detected in any of the samples from these tributaries.
- **Dutch Kills**. 2,3,7,8-TCDD was detected in one sample, at a concentration of 9.9 ng/kg.
- **Whale Creek**. 2,3,7,8-TCDD was detected in one sample, at a concentration of 2.4 ng/kg.

4.4.3.6 Cu

Cu concentrations in native material in each reach of the Study Area are shown in the box plot and longitudinal profile provided in Figures 4-84 and 4-85, respectively.
Cu concentrations in native material are generally one to two orders of magnitude lower than surface sediment within most of the Study Area (see Figure 4-84). Native material concentrations do not exhibit notable longitudinal patterns (see Figure 4-85). Finally, the median concentration of Cu in native material (22 mg/kg) is similar to regional geological concentrations (21 mg/kg, estimated as the median value of abundance in crust and soil from several sources; Mahler et al. 2006; NJDEP 1993; NYSDEC and NYSDOH 2006; Rudnick and Gao 2003; USGS 2016).

The range, arithmetic average, and median for each reach are as follows (no notable patterns were identified in any reach):

- **CM 0 – 1**. Cu concentrations range from 3.6 to 180 mg/kg (arithmetic average = 32 mg/kg; median = 24 mg/kg).
- **CM 1 – 2**. Cu concentrations range from 5.2 to 340 mg/kg (arithmetic average = 31 mg/kg; median = 17 mg/kg).
- **CM 2+**. Cu concentrations range from 6.1 to 14,000 mg/kg (arithmetic average = 260 mg/kg; median = 32 mg/kg). Elevated Cu in the Turning Basin was observed in one sample (at 14,000 mg/kg). All others are less than approximately 200 mg/kg (with a median value of approximately 30 mg/kg, as shown in Figure 4-84).
- **English Kills**. Cu concentrations range from 3.9 to 170 mg/kg (arithmetic average = 23 mg/kg; median = 13 mg/kg).
- **East Branch**. Cu concentrations range from 6.0 to 190 mg/kg (arithmetic average = 27 mg/kg; median = 14 mg/kg).
- **Maspeth Creek**. Cu concentrations range from 14 to 29 mg/kg (arithmetic average and median = 22 mg/kg).

- **Dutch Kills**. Cu concentrations range from 14 to 1,600 mg/kg (arithmetic average = 170 mg/kg; median = 24 mg/kg).
- **Whale Creek**. Cu concentrations range from 7.1 to 83 mg/kg (arithmetic average = 32 mg/kg; median = 27 mg/kg).

4.4.3.7 Pb

Pb concentrations in native material in each reach of the Study Area are shown in the box plot and longitudinal profile provided in Figures 4-88 and 4-89, respectively. Pb concentrations in native material are generally two to three orders of magnitude lower than surface sediment within most of the Study Area (see Figure 4-88). Native material concentrations do not exhibit notable longitudinal patterns (see Figure 4-89).

The range, arithmetic average, and median for each reach are as follows (no notable patterns were identified in any reach):

- **CM 0 – 1**. Pb concentrations range from 3.2 to 180 mg/kg (arithmetic average = 15 mg/kg; median = 7.5 mg/kg).
- **CM 1 – 2**. Pb concentrations range from 1.5 to 160 mg/kg (arithmetic average = 15 mg/kg; median = 6.5 mg/kg).
- **CM 2+**. Pb concentrations range from 1.5 to 1,900 mg/kg (arithmetic average = 42 mg/kg; median = 8.1 mg/kg). Elevated Pb in the Turning Basin was observed in one sample (at 1,900 mg/kg). All others are less than 100 mg/kg (as shown in Figure 4-89).
- **English Kills**. Pb concentrations range from 0.87 to 65 mg/kg (arithmetic average = 6.9 mg/kg; median = 3.2 mg/kg).
- **East Branch**. Pb concentrations range from 1.9 to 77 mg/kg (arithmetic average = 11 mg/kg; median = 6.0 mg/kg).
- **Maspeth Creek**. Pb concentrations range from 6.0 to 13 mg/kg (arithmetic average = 8.6 mg/kg; median = 7.8 mg/kg).
- **Dutch Kills**. Pb concentrations range from 4.9 to 1,200 mg/kg (arithmetic average = 130 mg/kg; median = 7.8 mg/kg). Elevated Pb was observed in one sample at 1,200 mg/kg and in another sample at 110 mg/kg; all others are close to (or less than) 10 mg/kg (as shown in Figure 4-89).

- **Whale Creek**. Pb concentrations range from 3.2 to 59 mg/kg (arithmetic average = 18 mg/kg; median = 8.8 mg/kg).

4.5 Sediment Traps

4.5.1 Sediment Trap Dataset

As described in Section 2.1.3.4, the Phase 2 sediment trap study was conducted at 30 locations in the Study Area for a 9-month period, starting in mid-June 2014 and extending through late March 2015 (see Figure 2-17a through d).[106] The program was designed to allow for the assessment of spatial and temporal variability in sediment composition, gross deposition fluxes, and chemical concentrations of sediment depositing in the Study Area. The sediment traps collected depositing solids that are likely derived from multiple sources. These sources may include point source discharges (e.g., overland flow, CSOs, industrial discharge, stormwater), local sediment resuspension (e.g., propwash from ship traffic), and the East River. Samples for mass deposition fluxes and physical properties were collected monthly during each deployment. Samples for chemical parameters were collected quarterly for three quarters, hereafter referred to as Quarter 1 (Q1), Quarter 2 (Q2), and Quarter 3 (Q3).[107] This section provides a discussion of sediment trap results, including spatial and temporal patterns for several select conventional parameters (Section 4.5.2), as well as for TPAH (17), TPAH (34), C19-C36, TPCB, 2,3,7,8-TCDD, Cu, and Pb (Sections 4.5.3.1 through 4.5.3.7, respectively).

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[106] Sediment traps consisted of a 6-inch-diameter cylindrical collection device mounted to a heavy plate that allowed them to be placed atop the sediment bed and collect depositing sediment. More details are provided in Section 9.3 of the Phase 2 FSAP Volume 2 (Anchor QEA 2014d) and its associated standard operating procedure (SOP NC-22 – Surface Sediment Sampling Using In-Creek Sediment Traps).

[107] Sediment traps were set for the following three quarterly deployment periods:

- Q1 – Quarter 1 (June 9 to 13 through September 15 to 24, 2014). Note that two traps (one each at the heads of English Kills and East Branch) were observed to be full after the first month of deployment and were sampled at that time. Data from these samples are denoted as "Q1-July."
- Q2 – Quarter 2 (September 15 to 24 through December 15 to 19, 2014)
- Q3 – Quarter 3 (December 15 to 19, 2014, through March 23 to 27, 2015, except at five locations due to missing/displaced traps)

4.5.2 Mass Flux and Characteristics of Solids in Sediment Traps

This section provides an overview of spatial and temporal patterns in gross sediment deposition (mass), percent fines, TOC, and percent solids, which are key characteristics in the evaluation of current sediment transport and chemical distribution patterns.

4.5.2.1 Gross Solids Deposition

Deposition mass flux (mass per unit area per unit time) calculated from the sediment trap data can provide qualitative insights about gross sediment deposition within the Study Area. These mass flux calculations are considered "gross" deposition rates, in that they provide an estimate of the total mass per unit area of suspended sediment particles that settled into a sediment trap over a given period. Although these deposition rates do not necessarily represent the long-term NSR in the creek bed,[108] they can be used qualitatively to help understand sedimentation in the Study Area (see Section 5.2.1 of Appendix G). As such, the sediment trap data provide an accepted means of qualitatively assessing the spatial and temporal variability in gross sediment deposition rates within the Study Area. Gross sediment deposition rates were calculated from sediment trap data as mass of dry matter per unit area per unit time (e.g., milligrams per square centimeters per day [$mg/cm^2/day$]).[109] A plan view map and longitudinal profile are provided in Figures 4-100 and 4-101, respectively, with summary statistics provided in Table 4-34. The following is an overall summary:

- Gross sediment deposition generally decreases from downstream to upstream (i.e., with increasing distance from the East River), with some variability between sampling quarters.
- Slightly higher gross deposition rates were consistently noted at a few locations, including one in Whale Creek (WC016ST in Q2 and Q3) and one in the main stem at

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[108] Not all settling particles intercepted by a sediment trap would otherwise reach the sediment surface, become incorporated into the bedded sediment, and contribute to long-term sediment bed NSRs. Other sediment bed processes including resuspension and bioturbation also contribute to the long-term disposition of settling particles. As such, the gross measurement from this sediment trap study may overrepresent the NSR.

[109] Gross solids deposition flux was calculated using the following process: 1) multiplying the total mass of wet sediment accumulated in the traps at each location (measured in the field at the time of sampling) by laboratory-measured percent solids to yield a dry mass; and 2) dividing by the cross-sectional area of the sediment trap and the trap deployment period, to yield mg/cm2/day.

CM 1.1 (NC111ST). Higher gross deposition fluxes were also observed in some of the individual samples from Maspeth Creek, English Kills, and East Branch.

- At 80% of the locations, the highest deposition was measured in either Q1 or Q3, indicating some temporal variability in deposition patterns.

These gross deposition fluxes are qualitatively compared to other LOEs (e.g., geochronology data) used to understand NSRs in the Study Area, as part of the sediment transport modeling effort (see Section 5.2.1 of Appendix G).

4.5.2.2 Percent Fines

A plan view map and longitudinal profile are provided in Figures 4-102 and 4-103, respectively, with summary statistics provided in Table 4-35. The following provides a summary of percent fines:

- The materials depositing within the sediment traps are generally fine-grained, which is consistent with the low-energy, depositional nature of the system. The percent fines in the sediment traps exceed 80 wt% in two-thirds of the samples (when the three quarters are considered together).
- Lower percent fines, often less than 60 wt%, were consistently observed at 8 of the 30 locations, which are mostly in tributaries (English Kills, East Branch, Maspeth Creek, and Whale Creek), but also include 3 locations within the main stem.
- The arithmetic average sediment trap percent fines by reach for all quarters combined ranges from 57 wt% to 89 wt%. Arithmetic average surface sediment percent fines concentrations by reach are generally similar and range from 43 wt% to 82 wt% (see Section 4.2.2.1). In the tributaries, these ranges are 43 wt% to 63 wt% for surface sediment and 57 wt% to 89 wt% for sediment traps. Although the arithmetic averages of percent fines for sediment traps in the tributaries are slightly higher than those in surface sediment, the sediment traps do show patterns of lower percent fines in the tributaries with CSOs (i.e., Dutch Kills, Maspeth Creek, East Branch, and English Kills) relative to the main stem. Except for Dutch Kills (arithmetic average: 89 wt%), the arithmetic average percent fines in tributaries with CSOs for sediment traps range from 58 wt% to 77 wt%. Arithmetic averages in the main stem range from 84 wt% to 87 wt%.

- In general, there was no consistent difference in percent fines observed among the three quarters.

4.5.2.3 Total Organic Carbon

A plan view map and longitudinal profile are provided in Figures 4-104 and 4-105, respectively, with summary statistics provided in Table 4-36. The following is a summary of TOC:

- The TOC content of the solids collected in the sediment traps was generally high as compared to other aquatic systems, consistent with the surface sediment (see Section 4.2.2.2); this reflects the depositional nature and high organic loads in the system. TOC for sediment trap samples in the Study Area ranged from 2.2 wt% (English Kills in Q1) to 16 wt% (Maspeth Creek in Q3), with an arithmetic average of 6.9 wt%.
- In general, the spatial distribution indicates TOC increases from downstream to upstream (except in Whale Creek, where TOC is relatively low). TOC results were generally higher in Maspeth Creek (arithmetic average = 10.6 wt%), English Kills (arithmetic average = 8.4 wt%), and East Branch (arithmetic average = 9.4 wt%) than in the rest of the Study Area (arithmetic average = 5.7 wt%).
- In general, differences among locations are likely due to the deposition of different combinations of local sediment and solids from the various sources (e.g., East River, point sources, propwash, and solids transported between reaches), and the sediment trap samples are interpreted to reflect a combination of these sources and processes that vary spatially and temporally. The overall higher TOC concentrations observed at the upstream locations (generally consistent with the presence of CSOs at the heads of the tributaries) may reflect differences in sources of solids (consistent with the pattern in the surface sediment percent fines and TOC data discussed in Section 4.2.2.3). At more than 80% of the locations, the highest TOC was measured in either Q2 or Q3, indicating some temporal variability in the composition of depositing solids. This temporal variation could be a result of a number of factors, including (but not limited to) differences in precipitation (which drives differences in point source flows from CSOs, stormwater, and other discharges) and vessel propwash. Further discussion is provided in Section 6.4.3.3.

4.5.2.4 Percent Solids

A plan view map and longitudinal profile are provided in Figures 4-106 and 4-107, respectively, with summary statistics provided in Table 4-37. Percent solids from the sediment trap samples steadily decrease from downstream to upstream, and within the main tributaries. This observed percent solids trend is generally the inverse of the observed TOC trend (see Section 4.5.2.3), as expected. That is, whereas percent solids are generally observed to decrease with CM (see Figure 4-107), TOC is observed to generally increase with CM (see Figure 4-105). This pattern likely reflects varied sources of solids across the Study Area (i.e., a gradual shift from lower TOC and higher solids near the mouth, to higher TOC and lower solids in the upper reaches). However, it is recognized that due to the various complexities of the processes by which sediments were collected by the traps and subsequently sampled, the percent solids data by themselves are not a strong indicator of the sources of solids.

4.5.3 Distribution of Contaminants

4.5.3.1 TPAH (17)

4.5.3.1.1 Spatial Patterns

TPAH (17) concentrations for sediment trap samples in the Study Area (the three quarterly sampling events combined) are summarized in Table 4-38; a plan view map and longitudinal profile showing the data are provided in Figures 4-108 and 4-109, respectively.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1 and CM 1 – 2**. TPAH (17) concentrations range from 5.3 to 48 mg/kg (arithmetic average = 17 mg/kg [CM 0 – 1] and 23 mg/kg [CM 1 – 2]; median = 14 mg/kg [CM 0 – 1] and 22 mg/kg [CM 1 – 2]) and are generally lower in this portion of the main stem (as compared to CM 2+), with the lowest values near the mouth.
- **CM 2+**. TPAH (17) concentrations range from 3.5 to 120 mg/kg (arithmetic average = 42 mg/kg; median = 41 mg/kg) and are generally highest in this portion of the main stem (as compared to CM 0 – 2), with the highest concentrations near the confluence of East Branch and English Kills.

- **English Kills**. TPAH (17) concentrations range from 5.7 to 400 mg/kg (arithmetic average = 110 mg/kg; median = 95 mg/kg). The highest discrete TPAH (17) sample result, 400 mg/kg, was observed in English Kills at CM 3.0 in Q2. Concentrations generally decrease from this location, both downstream in the main stem and upstream toward the head of the tributary.
- **East Branch**. TPAH (17) concentrations range from 5.6 to 170 mg/kg (arithmetic average = 59 mg/kg; median = 25 mg/kg). Concentrations are approximately 2 to 3 times higher downstream as compared to upstream.
- **Maspeth Creek**. TPAH (17) concentrations range from 9.1 to 110 mg/kg (arithmetic average = 53 mg/kg; median = 29 mg/kg). Concentrations at the three locations are generally similar to East Branch and those in the adjacent CM 2+.
- **Dutch Kills**. TPAH (17) concentrations range from 8.4 to 130 mg/kg (arithmetic average = 41 mg/kg; median = 23 mg/kg). Concentrations increase somewhat with distance upstream from its confluence with the main stem.
- **Whale Creek**. TPAH (17) concentrations range from 20 to 97 mg/kg (arithmetic average = 47 mg/kg; median = 24 mg/kg). Concentrations at the single sampling location are generally similar to those found nearby in the main stem.

4.5.3.1.2 Temporal Patterns

Table 4-38 also provides a summary of TPAH (17) data for each of the three quarterly sampling events. Arithmetic average TPAH (17) concentrations are 27 mg/kg in Q1, 75 mg/kg in Q2, and 37 mg/kg in Q3. Cross plots that compare TPAH (17) results at each location from Q2 versus Q1 (left panel), Q3 versus Q1 (middle panel), and Q3 versus Q2 (right panel) are presented in Figure 4-110. From these plots, and to some extent in Figure 4-109, a relatively consistent temporal pattern is evident. TPAH (17) concentrations from Q2 are consistently higher than in Q1 and Q3, and Q3 results are consistently higher than in Q1 at the majority of locations. One notable exception is in Whale Creek, where the concentration from the Q1 sample (97 mg/kg) is approximately 4 to 5 times higher than the concentrations from Q2 (20 mg/kg) and Q3 (24 mg/kg).

Section 6.4.3.3 contains interpretation of the sediment trap data for TPAH (17), including comparisons between surface sediment and point source concentrations and evaluation of potential factors that may have contributed to the observed temporal variability.

4.5.3.2 TPAH (34)

4.5.3.2.1 Spatial Patterns

TPAH (34) concentrations for sediment trap samples in the Study Area (the three quarterly sampling events combined) are summarized in Table 4-39; a plan view map and longitudinal profile showing the data are provided in Figures 4-111 and 4-112, respectively.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1 and CM 1 – 2**. TPAH (34) concentrations range from 9.6 to 88 mg/kg (arithmetic average = 26 mg/kg [CM 0 – 1] and 44 mg/kg [CM 1 – 2]; median = 24 mg/kg [CM 0 – 1] and 40 mg/kg [CM 1 – 2]) and are generally lower in this portion of the main stem (as compared to CM 2+), with the lowest values near the mouth.
- **CM 2+**. TPAH (34) concentrations range from 6.8 to 240 mg/kg (arithmetic average = 86 mg/kg; median = 84 mg/kg) and are generally highest in this portion of the main stem (as compared to CM 0 – 2), with the highest concentrations in the Turning Basin and near the confluence of East Branch and English Kills.
- **English Kills**. TPAH (34) concentrations range from 11 to 960 mg/kg (arithmetic average = 230 mg/kg; median = 220 mg/kg). The highest TPAH (34) sample result, 960 mg/kg, was observed in English Kills at CM 3.0 in Q2. Concentrations generally decrease from this location, both downstream in the main stem and upstream toward the head of the tributary.
- **East Branch**. TPAH (34) concentrations range from 9.7 to 370 mg/kg (arithmetic average = 120 mg/kg; median = 50 mg/kg). Downstream concentrations are approximately 2 to 3 times higher than upstream concentrations.
- **Maspeth Creek**. TPAH (34) concentrations range from 16 to 220 mg/kg (arithmetic average = 96 mg/kg; median = 56 mg/kg). Concentrations at the three locations are generally similar to East Branch and those in the adjacent CM 2+.

- **Dutch Kills**. TPAH (34) concentrations range from 15 to 230 mg/kg (arithmetic average = 72 mg/kg; median = 49 mg/kg). Concentrations increase somewhat with distance upstream from its confluence with the main stem.
- **Whale Creek**. TPAH (34) concentrations range from 70 to 340 mg/kg (arithmetic average = 160 mg/kg; median = 82 mg/kg). Concentrations at the single sampling location are generally similar to those found nearby in the main stem (other than the sample from Q1, which is higher).

4.5.3.2.2 Temporal Patterns

Table 4-39 provides a summary of TPAH (34) data for each of the three quarterly sampling events. Arithmetic average TPAH (34) concentrations are 55 mg/kg in Q1, 150 mg/kg in Q2, and 84 mg/kg in Q3. Figure 4-113 presents a series of cross plots that compare TPAH (34) results at each location between quarters. From these plots, and to some extent in Figure 4-112, a relatively consistent temporal pattern is evident. TPAH (34) concentrations from Q2 are consistently higher than in Q1 and Q3, and Q3 results are consistently higher than in Q1 at the majority of locations. One notable exception is in Whale Creek, where the concentration from the Q1 sample (340 mg/kg) is approximately 4 to 5 times higher than the concentrations from Q2 (70 mg/kg) and Q3 (82 mg/kg)

4.5.3.3 C19-C36

4.5.3.3.1 Spatial Patterns

C19-C36 concentrations for sediment trap samples in the Study Area (the three quarterly sampling events combined) are summarized in Table 4-40; a plan view map and longitudinal profile showing the data are provided in Figures 4-114 and 4-115, respectively. Note that not all sediment trap samples were analyzed for C19-C36; due to limited accumulation of material in traps for some locations and quarters, priority was given to PAHs, PCBs, metals, and TOC ahead of other analyte groups (see Section 4.3.1.2 of Appendix Bi and Table B9-6 of the Phase 2 FSAP Volume 2).

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1 and CM 1 – 2**. C19-C36 concentrations range from 21 to 1,600 mg/kg (arithmetic average = 92 mg/kg [CM 0 – 1] and 520 mg/kg [CM 1 – 2]; median = 49 mg/kg [CM 0 – 1] and 270 mg/kg [CM 1 – 2]) and generally increase going upstream from the East River.
- **CM 2+**. C19-C36 concentrations range from 170 to 1,700 mg/kg (arithmetic average = 790 mg/kg; median = 630 mg/kg) and are generally greater than concentrations in CM 1 – 2.
- **English Kills**. C19-C36 concentrations range from 1,300 to 9,800 mg/kg (arithmetic average = 4,900 mg/kg; median = 3,500 mg/kg) and are all higher than in the main stem.
- **East Branch**. C19-C36 concentrations range from 690 to 6,400 mg/kg (arithmetic average = 2,900 mg/kg; median = 2,700 mg/kg) and are higher than in the nearby main stem in three of the five samples, with the other two samples being in the range of the main stem.
- **Maspeth Creek**. C19-C36 concentrations range from 650 to 5,800 mg/kg (arithmetic average = 3,200 mg/kg; median = 2,600 mg/kg) and are higher than in the nearby main stem for six of the seven samples from this tributary.
- **Dutch Kills**. C19-C36 concentrations range from 24 to 2,000 mg/kg (arithmetic average = 770 mg/kg; median = 630 mg/kg). Concentrations increase somewhat with distance upstream from its confluence with the main stem.
- **Whale Creek**. C19-C36 concentrations range from 1,100 to 1,900 mg/kg (arithmetic average = 1,400 mg/kg; median = 1,100 mg/kg). Concentrations at the single sampling location are generally similar to those found nearby in the main stem.

4.5.3.3.2 Temporal Patterns

Table 4-40 provides a summary of C19-C36 data for each of the three quarterly sampling events. Arithmetic average C19-C36 concentrations are 880 mg/kg in Q1, 2,000 mg/kg in Q2, and 2,100 mg/kg in Q3. Figure 4-116 presents a series of cross plots that compare C19-C36 results at each location between quarters. While these plots indicate some variability, there does not appear to be any consistent temporal differences in C19-C36 concentrations among the three quarterly sampling events.

4.5.3.4 TPCB

4.5.3.4.1 Spatial Patterns

TPCB concentrations for sediment trap samples in the Study Area (the three quarterly sampling events combined) are summarized in in Table 4-41; a plan view map and longitudinal profile showing the data are provided in Figures 4-117 and 4-118, respectively.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1 and CM 1 – 2**. TPCB concentrations range from 0.41 to 1.6 mg/kg (arithmetic average = 0.53 mg/kg [CM 0 – 1] and 1.0 mg/kg [CM 1 – 2]; median = 0.51 mg/kg [CM 0 – 1] and 1.0 mg/kg [CM 1 – 2]) and are generally lower in this portion of the main stem (as compared to CM 2+), with the lowest concentrations near the East River and a slight increase moving upstream toward CM 2.
- **CM 2+**. TPCB concentrations range from 1.2 to 5.4 mg/kg (arithmetic average = 2.3 mg/kg; median = 1.7 mg/kg) and are generally highest in this portion of the main stem (as compared to CM 0 – 2), with the highest concentrations near the confluence of East Branch and English Kills.
- **English Kills**. TPCB concentrations range from 0.73 to 15 mg/kg (arithmetic average = 6.4 mg/kg; median = 4.4 mg/kg). Concentrations are generally higher in this tributary than in much of the main stem and decrease from CM 3.0 upstream toward the head of the tributary.
- **East Branch**. TPCB concentrations range from 0.13 to 2.2 mg/kg (arithmetic average = 0.94 mg/kg; median = 0.18 mg/kg). Concentrations at the more downstream location are approximately one order of magnitude higher than at the more upstream location in this tributary.
- **Maspeth Creek**. TPCB concentrations range from 0.25 to 28 mg/kg (arithmetic average = 4.4 mg/kg; median = 1.2 mg/kg). Concentrations are variable but are generally lower at the upstream location compared to the two locations at the confluence with the main stem; one elevated concentration (28 mg/kg) was observed at the head of the tributary in one of the quarters.
- **Dutch Kills**. TPCB concentrations range from 0.83 to 27 mg/kg (arithmetic average = 4.5 mg/kg; median = 1.4 mg/kg). Concentrations within the three sediment trap

locations are similar to one another and to those in locations in the adjacent main stem, in all but one case. The concentration in one sample collected near the mouth in Q2 (27 mg/kg) is more than an order of magnitude higher than the concentration of the other samples in this tributary (as well as most samples from the Study Area).

- **Whale Creek**. TPCB concentrations range from 1.1 to 2.5 mg/kg (arithmetic average = 1.8 mg/kg; median = 1.9 mg/kg). Concentrations at the single sampling location are slightly higher than those found nearby in the main stem.

4.5.3.4.2 Temporal Patterns

Table 4-41 provides a summary of TPCB data for each of the three quarterly sampling events. Arithmetic average TPCB concentrations are 1.8 mg/kg in Q1, 3.9 mg/kg in Q2, and 2.4 mg/kg in Q3. Figure 4-119 presents a series of cross plots that compare PCB results at each location between quarters. These plots do not indicate consistent differences in TPCB concentrations among the three quarterly sampling events, aside from the two elevated samples from Q2 (one in Maspeth Creek and one in Dutch Kills). Concentrations at a given location were generally similar (i.e., within a factor of 2 or less) among the three quarters sampled.

Section 6.4.3.3 contains interpretation of the sediment trap data for TPCB, including comparisons between surface sediment and point source concentrations and evaluation of potential factors that may have contributed to the observed temporal variability.

4.5.3.5 2,3,7,8-TCDD

4.5.3.5.1 Spatial Patterns

2,3,7,8-TCDD concentrations for sediment trap samples in the Study Area (the three quarterly sampling events combined) are summarized in in Table 4-42; a plan view map and longitudinal profile showing the data are provided in Figures 4-120 and 4-121, respectively. Similar to C19-C36 (see Section 4.5.3.3), some samples were not analyzed for 2,3,7,8-TCDD due to sample volume limitations.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1 and CM 1 – 2**. 2,3,7,8-TCDD concentrations range from 0.67 to 16 ng/kg (arithmetic average = 4.6 ng/kg [CM 0 – 1] and 6.9 ng/kg [CM 1 – 2]; median = 4.5 ng/kg [CM 0 – 1] and 6.9 ng/kg [CM 1 – 2])
- **CM 2+**. 2,3,7,8-TCDD concentrations range from 1.8 to 21 ng/kg (arithmetic average = 6.1 ng/kg; median = 5.3 ng/kg) and are generally similar to those in CM 0 – 2.
- **English Kills**. 2,3,7,8-TCDD concentrations range from 1.3 to 14 ng/kg (arithmetic average = 5.5 ng/kg; median = 2.6 ng/kg). Concentrations generally decrease from CM 3.0 upstream toward the head of the tributary.
- **East Branch**. 2,3,7,8-TCDD was not detected in any of the samples from this tributary.
- **Maspeth Creek**. 2,3,7,8-TCDD concentrations range from 0.51 to 3.1 ng/kg (arithmetic average = 2.2 ng/kg; median = 2.8 ng/kg). Concentrations are generally lower in this tributary compared to the main stem.
- **Dutch Kills**. 2,3,7,8-TCDD concentrations range from 1.8 to 12 ng/kg (arithmetic average = 4.4 ng/kg; median = 3.2 ng/kg). Concentrations in this tributary are highest near the confluence with the main stem.
- **Whale Creek**. 2,3,7,8-TCDD concentrations range from 8.3 to 18 ng/kg (arithmetic average and median = 13 ng/kg). Concentrations at the single sampling location are generally consistent with those found nearby in the main stem.

4.5.3.5.2 Temporal Patterns

Table 4-42 provides a summary of 2,3,7,8-TCDD data for each of the three quarterly sampling events. Arithmetic average 2,3,7,8-TCDD concentrations are 5.5 ng/kg in Q1, 7.0 ng/kg in Q2, and 6.5 ng/kg in Q3. Figure 4-122 presents a series of cross plots that compare 2,3,7,8-TCDD results at each location between quarters. While these plots do indicate some variability, there does not appear to be any consistent temporal differences in 2,3,7,8-TCDD concentrations among the three quarterly sampling events (unlike what was observed for TPAH [17], TPAH [34], TPCB, and Cu [in some cases]).

4.5.3.6 Cu

4.5.3.6.1 Spatial Patterns

Cu concentrations for sediment trap samples in the Study Area (the three quarterly sampling events combined) are summarized in Table 4-43; a plan view map and longitudinal profile showing the data are provided in Figures 4-123 and 4-124, respectively.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1 and CM 1 – 2**. Cu concentrations range from 75 to 500 mg/kg (arithmetic average = 130 mg/kg [CM 0 – 1] and 310 mg/kg [CM 1 – 2]; median = 120 mg/kg [CM 0 – 1] and 290 mg/kg [CM 1 – 2]) and generally increase going upstream from the East River.
- **CM 2+**. Cu concentrations range from 63 to 3,400 mg/kg (arithmetic average = 710 mg/kg; median = 410 mg/kg). Concentrations are highest in sediment trap samples at CM 2.4 and CM 2.6, with three samples having concentrations greater than 1,000 mg/kg. The highest Cu sample result (3,400 mg/kg) was observed in CM 2+ in Q3.
- **English Kills**. Cu concentrations range from 74 to 3,400 mg/kg (arithmetic average = 800 mg/kg; median = 420 mg/kg). Concentrations at the downstream end of English Kills are somewhat higher than the rest of the Study Area. Cu averages 1,800 mg/kg at the downstream location and 310 mg/kg at the upstream location.
- **East Branch**. Cu concentrations range from 20 to 210 mg/kg (arithmetic average = 100 mg/kg; median = 85 mg/kg). Concentrations at the two locations are generally lower than much of the Study Area. The arithmetic average Cu concentration is 67 mg/kg at the downstream location and 130 mg/kg at the upstream location.
- **Maspeth Creek**. Cu concentrations range from 55 to 670 mg/kg (arithmetic average = 310 mg/kg; median = 260 mg/kg). The arithmetic average Cu concentrations are 430 mg/kg at the two downstream locations and 120 mg/kg at the upstream location.
- **Dutch Kills**. Cu concentrations range from 130 to 470 mg/kg (arithmetic average = 260 mg/kg; median = 300 mg/kg). Concentrations within Dutch Kills are all relatively similar to one another and to samples in the adjacent main stem.
- **Whale Creek**. Cu concentrations range from 260 to 1,400 mg/kg (arithmetic average = 660 mg/kg; median = 340 mg/kg). Concentrations at the single sampling location are higher than those found nearby in the main stem.

4.5.3.6.2 Temporal Patterns

Table 4-43 provides a summary of Cu data for each of the three quarterly sampling events. Arithmetic average Cu concentrations are 490 mg/kg in Q1, 370 mg/kg in Q2, and 490 mg/kg in Q3. Figure 4-125 presents a series of cross plots that compare Cu results at each location between quarters. In general, temporal variability is limited across most of the Study Area, with noted exceptions in English Kills and East Branch. At each of the four English Kills sediment trap locations, concentrations vary by 500 mg/kg or greater across the three quarterly sampling events. At the two East Branch sediment trap locations, concentrations vary by 100 mg/kg or more across sampling events.

Section 6.4.3.3 contains interpretation of the sediment trap data for Cu, including comparisons between surface sediment and point source concentrations and evaluation of potential factors that may have contributed to the observed temporal variability.

4.5.3.7 Pb

4.5.3.7.1 Spatial Patterns

Pb concentrations for sediment trap samples in the Study Area (the three quarterly sampling events combined) are summarized in Table 4-44; a plan view map and longitudinal profile showing the data are provided in Figures 4-126 and 4-127, respectively.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1 and CM 1 – 2**. Pb concentrations range from 72 to 340 mg/kg (arithmetic average = 140 mg/kg [CM 0 – 1] and 230 mg/kg [CM 1 – 2]; median = 140 mg/kg [CM 0 – 1] and 230 mg/kg [CM 1 – 2]) and increase somewhat going upstream from the East River.
- **CM 2+**. Pb concentrations range from 34 to 860 mg/kg (arithmetic average = 270 mg/kg; median = 220 mg/kg). The maximum concentration in the main stem was observed at CM 2.6 in Q1.
- **English Kills**. Pb concentrations range from 54 to 1,700 mg/kg (arithmetic average = 500 mg/kg; median = 400 mg/kg). Concentrations generally decrease from CM 3.0 upstream toward the head of the tributary.

- **East Branch**. Pb concentrations range from 20 to 430 mg/kg (arithmetic average = 150 mg/kg; median = 130 mg/kg). Concentrations at the two locations are generally lower than much of the Study Area.
- **Maspeth Creek**. Pb concentrations range from 33 to 370 mg/kg (arithmetic average = 190 mg/kg; median = 170 mg/kg). Concentrations generally decrease from the confluence with the main stem upstream toward the head of the tributary.
- **Dutch Kills**. Pb concentrations range from 100 to 310 mg/kg (arithmetic average = 220 mg/kg; median = 240 mg/kg). Concentrations within Dutch Kills are all relatively similar to one another and to samples in the adjacent main stem.
- **Whale Creek**. Pb concentrations range from 210 to 1,100 mg/kg (arithmetic average = 530 mg/kg [the highest among Study Area reaches]; median = 270 mg/kg). At the single sampling location, the concentration from Q1 is higher than those found nearby in the main stem, and the concentrations from Q2 and Q3 are generally consistent with those found nearby in the main stem.

4.5.3.7.2 Temporal Patterns

Table 4-44 provides a summary of Pb data for each of the three quarterly sampling events. Arithmetic average Pb concentrations are 295 mg/kg in Q1, 201 mg/kg in Q2, and 230 mg/kg in Q3. Figure 4-128 presents a series of cross plots that compare Pb results at each location between quarters. In general, samples collected from CM 0 – 1 and CM 1 – 2 are most consistent across the three sampling quarters, with greater variability observed in the other reaches. The data from Q1 were higher than those from Q2 for two-thirds of the locations; fewer differences were observed for the other comparisons among quarters.

4.6 NAPL

4.6.1 NAPL Dataset and Evaluation Approach

4.6.1.1 Overview

During the RI, NAPL presence or absence was identified by direct visual observation of sediment and native material in surface sediment grabs and cores and by performing shake tests on sediment or native material samples and visually observing if NAPL separated from the sediment or native material (discussed in Section 4.6.1.2).

The NAPL observational datasets used in the RI are described in detail in Section 2 of Appendix C and are summarized as follows:

- During Phase 1, the presence of NAPL in sediment and underlying native material was confirmed at four locations in the Study Area.
- During Phase 2, data were collected to evaluate and delineate the extent of NAPL in sediment and native material.
- During Part 1 of the FS, additional data were collected to refine the delineation of NAPL in sediment and native material.
  - In addition to the FS NAPL investigation, visual observations of sheen and NAPL were documented for surface sediment grabs and cores collected as part of other FS Part 1 programs.
- In addition to NAPL observational data collected during the RI, previous NAPL observational data collected by National Grid (GEI 2009b, 2012, and 2017) were reviewed, determined to be acceptable, and subsequently approved by USEPA for use in the RI (Anchor QEA 2014b).

As part of the Phase 2 program, a core logging procedure was developed for identifying and confirming the presence or absence of NAPL in sediment and native material. The Phase 2 core logging procedure included the following two steps (Anchor QEA 2014b, 2014d):

1. The visual observations of potential NAPL presence or absence in cores were recorded during core processing.[110]
2. The visual observations of potential NAPL presence or absence in cores were confirmed by a shake test.[111]

"Potential NAPL" refers to visual observations of possible indicators of NAPL, including sheen (iridescent color on the surface of water or core sample), blebs (discrete NAPL droplets), NAPL coating on sediment particles, or apparent NAPL saturation in sediment

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[110] Upon collection of a surface sediment grab or core, the grab or core was opened, and a systematic description of the sediment was recorded (i.e., sediment type [lithology], density/consistency, moisture content, color, structure, and anthropogenic material [if present]), followed by any odors or visual observations of potential sheen or NAPL (if present).

[111] A shake test consists of placement of sediment and distilled water into a clean laboratory jar, which is shaken and allowed to equilibrate, to observe whether separate phase liquid separates out.

pore spaces. Visual observations were described as "potential sheen" or "potential NAPL" until sheen or NAPL presence was confirmed by a shake test. The Phase 2 NAPL identification methods, including confirming the presence or absence of NAPL using shake tests (Anchor QEA 2014b, 2014d), are described in detail in Section 2.2.1 of Appendix C.

The FS NAPL distribution refinement program, performed during Part 1 of the FS, collected data to refine the delineation of three limited areas where a relatively greater magnitude of NAPL was observed (i.e., Category 2/3 Areas, discussed further in Section 4.6.1.3) based on the results of Phase 2. The Phase 2 NAPL identification methods were used during logging of FS NAPL distribution refinement cores.

4.6.1.2 NAPL Dataset

As described in Section 4.6.1.1, the NAPL observational data used to delineate the presence and extent of NAPL in this RI Report are a compilation of several different datasets, including data from Phase 1 sampling, data from Phase 2 sampling, data from Part 1 of the FS, and data collected by others that have been approved by USEPA for use in the RI. Visual observations and shake test results were classified similarly across these sampling programs using standardized terminology.

Visual observations of potential NAPL presence or absence in sediment and native material were described in the following terms, consistent with the Phase 2 FSAP Volume 2 (Anchor QEA 2014d) and FS FSAP (Anchor QEA 2017c):

- No visual evidence of NAPL: No sheen or NAPL is observed in the sediment.
- Sheen: A sheen is present on a portion of the surface of the sediment; however, NAPL is not observed.

- Blebs: Discrete droplets of NAPL are present in the sediment, but for the most part, the sediment is not visually impacted. Typically, this is indicative of residual NAPL.[112]
- Coated: Sediment grains are coated with NAPL. There is not sufficient NAPL present to fully saturate the pore spaces.
- Saturated: The sediment pore spaces are filled with NAPL.

Shake tests performed during Phase 2 of the RI and Part 1 of the FS were performed under controlled conditions using the standardized method specified in the Phase 2 FSAP Volume 2 (Anchor QEA 2014d). The method consisted of adding uniform amounts of sediment and water to a clear, 2-ounce polystyrene jar, inverting the jar repeatedly for 10 seconds to suspend the sediment, and allowing the jar contents to equilibrate for 10 minutes. The jar and jar contents were then observed for the presence of a sheen, NAPL blebs, or a NAPL layer. Shaking the shake test jar mechanically separates the NAPL from the sediment, causing the NAPL to accumulate on the shake test jar walls, or (if a sufficient quantity of NAPL is present) as a layer on the surface of the water in the shake test jar. Because NAPL that may be residual in the sediment sample will accumulate after being agitated in a jar with water (i.e., the sediment matrix is disrupted by the agitation), shake testing provides a conservative indicator of the presence of NAPL in a sediment sample. Shake test results were logged and photographed.

Shake tests were performed on sediment and native material, where sheen or visual indicators of potential NAPL were observed, to confirm whether or not NAPL was present in the sediment or native material. Where visual indicators of potential NAPL were not observed, one or more depth intervals were chosen for shake tests based on other criteria (e.g., photoionization detector readings, odor, or changes in sediment type). Visual

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[112] Residual NAPL is the condition where NAPL saturation is sufficiently low that the NAPL consists of discontinuous blebs trapped by capillary forces, so it is immobile. This classification is specific to the ability of the NAPL to advect (i.e., flow) as a nonaqueous fluid phase. The interpretation that blebs represent residual, immobile NAPL is based on the observation that, in core samples, the blebs are present as small, discontinuous droplets; this matches the description of residual NAPL as documented in the literature (Schwille 1988; Cohen and Mercer 1993; Pankow and Cherry 1996; API 2003; ITRC 2004; Sale et al. 2008; ITRC 2009; Kueper and Davies 2009). This language will be used hereafter, unless FS Part 1 NAPL mobility testing results are available and indicate that the NAPL is mobile. FS Part 1 NAPL mobility testing was performed for CM 0 – 2, as discussed in Section 4.4 of Appendix C (see also Section 6.4.7). Based on the results of that work, NAPL is immobile (i.e., residual) in the CM 0 – 2 reach. NAPL mobility testing for CM 2+ is being performed during Part 2 of the FS.

observations and shake test results were recorded for all primary cores collected during Phase 2 with a few exceptions,[113] and for all FS NAPL distribution refinement cores.

Shake test results were described in the following terms, consistent with the Phase 2 FSAP Volume 2 (Anchor QEA 2014d):

- Negative shake test result: No sheen or NAPL is observed in the shake test jar. A negative shake test result indicates that sheen and NAPL are not present in the sample tested.
- Shake test sheen result: A sheen is present on the surface of the water, but NAPL blebs or a NAPL layer is not observed. Per the Phase 2 RI Work Plan Volume 2 (Anchor QEA 2014b) and FS FSAP (Anchor QEA 2017c), a shake test sheen result indicates that NAPL is not present in the sample tested.
- Shake test blebs result: Discrete droplets of NAPL are present on the sidewalls of the shake test jar, on the water's surface, suspended in the water, or settled on the sediment surface submerged under the water in the jar. The degree of bleb accumulation (see Section 2.2.1.3 of Appendix C for more details) is noted on the field logs. A shake test blebs result indicates that residual NAPL is present in the sample tested.
- Shake test layer result: NAPL appears as a distinct layer within the shake test jar. Typically, when a NAPL layer was observed, the shake test jar walls were also coated in NAPL. A shake test layer result indicates that NAPL is present in the sample tested.

The two-step Phase 2 process used to screen sediment and native material for the presence or absence of NAPL (i.e., visual observations confirmed by shake tests) was performed on 200 cores located throughout the Study Area, whose locations are shown in Figure 4-129. Visual observations and shake test results were recorded on field logs (see Attachments Bi-C04 and Bi-C09 of Appendix Bi). Photographs of the cores and the shake test jars are provided in Attachments Bi-D04 and Bi-D09 of Appendix Bi. See Attachment C-E of Appendix C for photographs of cores with examples of visual observations of sheen, blebs, and coated and saturated visual observations. Photographs of select shake test jars, demonstrating examples

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[113] Visual observations and shake test results were recorded for all primary cores collected during Phase 2 with the exception of four cores, for which only visual observations were recorded. The four cores contained no visual evidence of potential NAPL, and the deviation from Phase 2 field methods was documented in QAPP Deviation Forms 5-3, 5-5, 5-10, and 6-3 (see Appendix Bi).

of negative, sheen, NAPL bleb, and NAPL layer results, are presented in Attachment C-F of Appendix C.

In addition to cores collected specifically to investigate the presence or absence of NAPL, 194 Phase 2 surface sediment grabs, 43 FS Part 1 surface sediment grabs, and 13 FS Part 1 cores (see Figure 4-130) were collected to support other programs. These surface sediment grabs and cores were processed for visual observations of potential sheen and NAPL presence or absence only. Visual observations were recorded using Phase 2 terminology; shake test confirmation of visual observations was not performed. Phase 2 and FS Part 1 surface sediment grab and core visual observations were used as a supplement to the observations from shake-tested Phase 2 and FS NAPL distribution refinement cores.

As described in Appendix C (see Section 2.4), historical data from other studies collected within the Study Area, but outside of the RI/FS process, were reviewed and evaluated for data usability to supplement the RI. Based on the results of that data usability review, 42 cores (see Figure 4-131) collected by National Grid were determined to be acceptable and were subsequently approved by USEPA for use in the RI dataset. The methods and terms used to classify visual observations in National Grid cores were consistent with the Phase 2 methods and terms for classifying visual observations; however, shake testing to confirm the presence or absence of NAPL was not performed. Per the Phase 2 FSAP Volume 2, in the absence of a shake test to confirm presence, National Grid data represent visual observations of potential NAPL. The National Grid data were used as a supplement to the observations from shake-tested Phase 2 and FS NAPL distribution refinement cores.

In total, 200 shake-tested cores (from the Phase 1, Phase 2, and FS Part 1 programs) were used to evaluate the nature and extent of NAPL present in Study Area surface and subsurface sediment and native material. These data were supplemented by 237 surface sediment grabs (from the Phase 2 and FS Part 1 programs) and 55 cores (from the FS Part 1 programs and National Grid investigations) that were processed for observations of potential sheen and NAPL. The most notable visual and shake test observations, over all depths observed at each core location in sediment and native material, are shown in Figures 4-132 and 4-133, respectively. The distribution of the most notable sheen and NAPL observations is described in detail in Section 3.3 of Appendix C.

In addition to these data, 239 sediment trap samples were collected, as discussed in Section 4.5.1. The sediment trap samples were processed for visual observations of potential sheen and NAPL presence or absence using Phase 2 terminology. A shake test confirmation of visual observations was not performed. The sediment trap visual observations, while not used in the NAPL evaluation (which focused on characterizing the nature and extent of NAPL in Study Area sediment and native material), are presented and discussed in Section 4.6.5 for completeness.

4.6.1.3 Evaluation Approach

To aid in data evaluation and to easily identify areas of more significant NAPL presence, cores were sorted into three groups, depending on the classification of visual observations and the shake test results. Typically, visual observation of NAPL blebs produced a shake test bleb result, and visual observations of coated or saturated NAPL produced a shake test layer result. A flow chart describing the process to identify the magnitude of NAPL observations is presented in Figure 4-134, and the following is a summary of the three groups into which cores were sorted and their associated NAPL observations:

- **Category 1A** cores contained no NAPL. Cores with negative or sheen shake test results were assigned to this category.
  - Of the 200 Phase 2 and FS Part 1 cores shake tested, the majority (110/200) were Category 1A cores (see Figure 4-135).
  - Of the 110 Category 1A cores, sheen was observed in surface sediment in 35% of the cores (38/110).
  - Of the 110 Category 1A cores, sheen was observed in subsurface sediment in the majority of the cores (81/110).
- **Category 1B** cores contained discrete, residual NAPL. Cores with bleb shake test results were assigned to this category. The Category 1B cores were further ranked by magnitude based on the degree to which blebs accumulated in the shake test.
  - Of the 200 cores shake tested, 32% (64/200) were Category 1B cores. Note that a combined 174/200 cores (87% of the shake-tested cores) contained either no observed NAPL or residual NAPL.
  - A targeted evaluation of Category 1B cores was performed to confirm that the NAPL observed was not associated with a more significant area of NAPL presence.

  Except for two Category 1B cores located in CM 2+, Category 1B cores were found to be not associated with areas where Category 2/3 NAPL is located (see Figure 4-135). A detailed summary of the Category 1B evaluation and results is presented in Section 5.3 of Appendix C.

- **Category 2/3** cores contained relatively more NAPL than was observed in Category 1B cores. Cores with a shake test layer were assigned to this category. Category 2/3 cores were generally associated with visual observations of coated or saturated NAPL in sediment or native material.
  - Of the 200 Phase 2 cores shake tested, 13% (26/200) were Category 2/3 cores.
  - The four Phase 1 cores that were shake-tested during the Phase 1 program and that had shake test layer results were conservatively classified as Category 3.

During the initial evaluation of NAPL observations, Category 2 and Category 3 cores were differentiated based on the type of visual observation for the sediment (or native material) that produced the shake test layer result. For Category 2 cores, the material that generated a shake test layer result had a visual observation of blebs. For Category 3 cores, the material that generated a shake test layer result had a visual observation of coated or saturated. One Category 2 core was identified; based on the layer shake test result, this Category 2 core was conservatively combined with the Category 3 locations, which also produced layers during shake testing. Collectively, the combined cores are referred to as Category 2/3 cores.

In general, visual observations were consistent with shake test results (i.e., visual observations that indicated the potential for relatively more NAPL to be present in sediment and native material were verified by shake test results [see Table C3-6 in Appendix C]). For those RI, FS Part 1, and National Grid visual observations that were not shake tested, surface sediment grabs and cores with a most notable visual observation of blebs were treated as Category 1B locations, and surface sediment grabs and cores with a most notable visual observation of coated or saturated were treated as Category 2/3 locations. For example, using this approach, 23 of the 42 National Grid cores were treated as Category 2/3 cores.

A detailed evaluation was performed to identify the extent of Category 2/3 NAPL observations and is presented in Section 5.4 of Appendix C.

The spatial distribution of NAPL in the Study Area is summarized in Sections 4.6.2 through 4.6.4 (see also Sections 3.3.1 through 3.3.3 and Section 5 of Appendix C for a detailed discussion).

4.6.2 Surface Sediment

NAPL was generally not observed in surface sediment (defined as the top 15 cm [6 inches] of sediment; see Section 4.2.1), as indicated by the distribution of observations shown in Figure 4-136. Observations of NAPL in surface sediment were limited to 11 locations—9 in the upstream portion of CM 2+ in the Turning Basin, 1 farther downstream in CM 2+, and 1 in the lower portion of English Kills.

Sheen was observed intermittently (i.e., located sporadically throughout an area, not clustered at a particular location) in surface sediments from CM 0 – 2, Dutch Kills, Whale Creek, East Branch, and Maspeth Creek, with no clear spatial pattern.

From CM 2 to 2.35, NAPL was not observed, but sheen was increasingly observed in surface sediments in the upstream direction. From CM 2.35 to 2.5, sheen was observed in many surface sediment locations, and potential NAPL blebs were observed in surface sediment at GPEC-SED23. Upstream of CM 2.5, sheen was observed in many surface sediment locations, and potential NAPL was observed in surface sediments at nine locations scattered throughout the reach. NAPL observations generally consisted of blebs; however, more notable observations of coated and saturated were observed in surface sediments at two locations in the southwest corner of the Turning Basin. These NAPL observations were evaluated as part of the Turning Basin Category 2/3 Area (see Section 4.6.3.2).

Sheen was frequently observed in surface sediments in the lower and middle portions of English Kills. Blebs were observed in surface sediment at one English Kills location near the cove at the lower bend in the creek. These NAPL observations were evaluated as part of the Lower English Kills Category 2/3 Area (see Section 4.6.3.3) to help define the Category 2/3 Area boundary. From the middle portion of the tributary to the head of English Kills, sheen observations decreased, and more than half of the surface sediments contained no visual observations.

4.6.3 Subsurface Sediment

Sheens were observed in subsurface sediment (defined as sediment from below 15 cm [6 inches]) to the sediment/native material interface) in Dutch Kills and Whale Creek; however, NAPL was not observed in subsurface sediment in these reaches (see Figure 4-137).

Blebs and sheen were observed in subsurface sediment in the main stem, Maspeth Creek, East Branch, and English Kills (see Figure 4-137). Bleb and sheen observations in cores and shake tests from the approximately lower 2 miles of the main stem, Maspeth Creek, and East Branch lacked a clear pattern in spatial and vertical distribution (see Section 5.3 of Appendix C). Bleb observations in the cores and shake tests were evaluated as part of the Category 1B evaluation, which confirmed that the NAPL observations in the majority of these areas were limited to blebs or sheens and were not laterally or vertically extensive—with the exception of CM 1.7, which was further evaluated as a Category 2/3 Area. The Category 1B evaluation process is described in more detail in Sections 5.1 and 5.3 of Appendix C.

Shake test layer results, and coated and saturated visual observations, were limited to the three Category 2/3 areas (see Figure 4-135), which are discussed in Sections 4.6.3.1 through 4.6.3.3. The Category 2/3 evaluation delineated the extent of Category 2/3 NAPL observations (i.e., identifying the lateral and vertical limits of the Category 2/3 NAPL) in sediment and native material and is described in more detail in Sections 5.1 and 5.4 of Appendix C. A boundary depicting the lateral extent of Category 2/3 NAPL in surface and subsurface sediment was developed based on the conservative assumption that Category 2/3 NAPL extended to the next closest core without Category 2/3 NAPL (i.e., the boundaries of Category 2/3 NAPL are generated by connecting the closest non-Category 2/3 cores) and to the shoreline.

4.6.3.1 Creek Mile 1.7 Category 2/3 Area

The CM 1.7 Category 2/3 NAPL Area was defined by two Phase 1 shake-tested cores and one Phase 2 core (see Figures 4-135 and 4-137). At these locations, NAPL that produced shake test layer results (i.e., Category 2/3 NAPL) was observed in thin discontinuous sand beds (1 to 3 cm [less than and up to slightly greater than 1 inch] thick, at a depth of 6 to 8 feet below the mudline, depending on the location) in the subsurface sediment and at the sediment/native material interface. The three cores were located within a 250-foot section of the creek clustered at CM 1.65; they were collocated with and surrounded by cores where Category 2/3 NAPL was not observed. FS NAPL distribution refinement cores (taken during Part 1 of the FS) were collected from eight additional stations to refine the lateral extent of NAPL observations in subsurface sediment and at the sediment/native material interface at CM 1.7. As discussed in Section 5.4.1.2 of Appendix C, Category 2/3 NAPL was not observed in any of the FS NAPL distribution refinement cores. However, NAPL that produced shake test bleb results (i.e., Category 1B NAPL) was observed intermittently several feet below the mudline, typically in intervals of less than 1.5 feet thick, though at two locations in intervals from 2.4 to 2.7 feet thick. Core intervals that produced shake test sheen results (i.e., Category 1A sheen) were observed intermittently in intervals ranging from 3 to 189 cm (0.1 to 6.2 feet) thick. Cross-sections depicting visual observations and shake test results in this area are presented in Figures C5-12 through C5-13f of Appendix C.

Figure 4-138 shows the approximate lateral extent of Category 2/3 NAPL observed in subsurface sediment in CM 1.7 (Category 2/3 NAPL was not observed in surface sediment or native material in this area). Two of the three Category 2/3 cores in this area are collocated with cores that do not contain Category 2/3 NAPL, indicating that even within the approximate extent of Category 2/3 NAPL, the Category 2/3 NAPL is discontinuous.

Note that Part 1 of the FS included NAPL mobility testing of 24 sediment samples and 4 native material samples from CM 0 – 2, including 7 sediment samples and 1 native material sample from the CM 1.7 Category 2/3 Area. Testing was performed on samples with the highest visible indication of NAPL presence based on core photography, representing the worst-case conditions. Of the 24 sediment samples and 4 native material samples tested from CM 0 – 2, none exhibited NAPL mobility (i.e., the NAPL was immobile in these samples) during Stage 1 centrifuge testing. Results of the FS Part 1 NAPL mobility testing are discussed in detail in Section 6.4.7.2 (see also Section 4 of Appendix C).

4.6.3.2 Turning Basin Category 2/3 Area

The Turning Basin Category 2/3 NAPL observations (visual observations of coated or saturated, shake test layer results) were generally limited to cores located in the west and southwest portions of the Turning Basin (see Figures 4-135 and 4-137). The thickest intervals of visual observations of Category 2/3 NAPL in subsurface sediment in this area were primarily observed in the southwest corner of the Turning Basin, where observations ranged from 10 to 305 cm (0.3 to 10 feet) thick. Moving downstream along the west perimeter of the Turning Basin, Category 2/3 NAPL observations in sediment decreased to thicknesses of less than 1 foot and tended to be located at (or near) the sediment/native material interface. FS NAPL distribution cores were collected from two stations in the Turning Basin to further refine the lateral extent of Category 2/3 NAPL in native material and are discussed in Section 4.6.4. In addition, Category 1B NAPL (shake test bleb results and visual observations of blebs) was observed in subsurface sediments along the Brooklyn shoreline. Category 1A sheen (shake test sheen results and visual observations of sheen) was observed in subsurface sediment at most sample locations, in intervals ranging from 10 to 305 cm (0.3 to 10 feet) thick. Cross-sections depicting visual observations and shake test results in this area are presented in Figures C5-15 through C-16l of Appendix C.

Figure 4-139 shows the approximate lateral extent of Category 2/3 NAPL observed in surface and subsurface sediment in the Turning Basin Area. Category 2/3 NAPL in sediment was observed at various depths and lengths, ranging from the mudline (two cores) to 22 feet below the mudline.

4.6.3.3 Lower English Kills Category 2/3 Area

NAPL in subsurface sediment in this Category 2/3 Area was generally limited to the lower bend of English Kills, 1,300 feet upstream from the confluence with East Branch (see Figures 4-135 and 4-137). NAPL was generally present in thin (less than 3 cm [less than 1 inch]) intervals. Most of these thin intervals were located at or near the sediment/native material interface, except for one location, where blebs were observed visually and/or in shake tests in surface and subsurface sediment from the mudline down to the sediment/native material interface at 3.5 feet below the mudline. FS NAPL distribution refinement cores were collected from 15 additional stations to refine the lateral extent of NAPL observations in sediment and native material in English Kills. As discussed in Section 5.4.2.1 of Appendix C, Category 2/3 NAPL in subsurface sediment is limited to five cores (EK005BSC, EK006SC-D, EK025SC-H,I, EK093SC-A, and EK104SC-A). These locations are surrounded by cores where Category 2/3 NAPL was not observed in subsurface sediment.

Core intervals that produced shake test sheen results (i.e., Category 1A sheen) were observed in the subsurface sediment at a number of sampling locations in this area in intervals typically 3 to 4 feet thick, but at one location, the interval containing sheen was 10 feet thick. Cross-sections depicting visual observations and shake test results in this area are presented in Figures C5-19 through C5-20m of Appendix C.

Figure 4-140 shows the approximate lateral extent of Category 2/3 NAPL observed in subsurface sediment in the Lower English Kills Area (Category 2/3 NAPL was not observed in surface sediment). Category 2/3 NAPL was observed to a limited degree in sediment, in an area located within the larger footprint of Category 2/3 NAPL observations in native material (see Section 4.6.4). Where Category 2/3 NAPL was observed in sediment, it was generally in thin intervals (1 to 8 cm [less than 1 to approximately 3 inches]) at various depths, 3 to 10 feet below the mudline.

4.6.4 Native Material

In native material, sheen and NAPL observations were generally limited to the Turning Basin and Lower English Kills Category 2/3 Areas (see Figure 4-133). Sheen and NAPL were not observed in native material from Dutch Kills, Whale Creek, or East Branch. Isolated sheens were observed in native material at several locations in the main stem (generally between CM 1.3 and 2) and at one location in Maspeth Creek. Except for one location where blebs (i.e., Category 1B NAPL) were observed in native material near CM 1.7 (see Figure 4-133), NAPL was not observed in native material in the main stem or Maspeth Creek.

In CM 2+, Category 1A sheen (shake test sheen results and sheen visual observations), as well as Category 1B NAPL (shake test bleb results and bleb visual observations) and Category 2/3 NAPL (shake test layer results and coated and/or saturated visual observations) were observed. These observations of sheen and NAPL were at or near the sediment/native material interface and extended into the native material, with the frequency and thickness of sheen and NAPL observations increasing in the upstream direction. Observations of NAPL in native material were primarily limited to the southwest portion of the CM2+ area along the shoreline, in the Turning Basin Category 2/3 Area (see Figure 4-133).

In the Turning Basin Category 2/3 Area, visual observations of potential NAPL and positive shake test results were observed in the following two intervals within the native material:

- The first interval of observations was present at, and within 10 feet of, the sediment/native material interface.
- The second interval of observations was present at greater depths, separated from the shallower native material observations by 7 to 28.5 feet of native material that had no observed NAPL.

Visual observations of NAPL in the deep native material were observed in three cores in the most southwestern corner of the Turning Basin (GPEC-SB112, GPEC-SB110, and GPEC-GT12; see Section 5.4.2.1 of Appendix C). FS NAPL distribution refinement cores were collected at two stations (NC374SC and NC375SC) to further refine NAPL in native material in the Turning Basin. As discussed in Section 5.4.2.1 of Appendix C, NAPL was only observed in one of the FS NAPL distribution refinement cores (NC374SC). NC374SC was collected to delineate the distribution of Category 2/3 NAPL at GPEC-SB112, where Category 2/3 NAPL was observed in deep native material. NC374SC penetrated 8 feet below the deepest Category 2/3 NAPL observation at GPEC-SB112; NAPL was not observed at NC374SC in deep native material. The results of FS core NC374SC indicate that the lateral extent of NAPL observed in deep native material at GPEC-SB112 is limited. With the additional data provided by NC374SC, Category 2/3 NAPL is delineated in the upstream, downstream, and offshore directions; see Figures C5-16d and C5-16i of Appendix C. Surrounding cores that penetrated to similar depths as NC374SC and GPEC-SB112 contained no visually observed NAPL, indicating that the deep native material impacts in the Turning Basin are localized and discontinuous. Figure 4-141 shows the approximate lateral extent of Category 2/3 NAPL observed in native material. Category 2/3 NAPL in native material was observed in two discrete areas within the footprint of the Category 2/3 NAPL in the sediment in the Turning Basin (see Section 4.6.3.2). The NAPL was primarily located in the upper 20 feet of native material, with discontinuous NAPL in deeper native material at three core locations.

In English Kills, observed NAPL in native material was generally limited to the lower portion of the tributary, where Category 1A sheen (shake test sheen results and sheen visual observations), Category 1B NAPL (shake test bleb results and bleb visual observations), and Category 2/3 NAPL (shake test layer results and coated and saturated visual observations) were observed at or near the sediment/native material interface and extending into the native material. Category 2/3 NAPL observations in native material were limited to a discrete area from approximately CM 2.95 to CM 3.1, in the Lower English Kills Category 2/3 Area (see Figure 4-133). The frequency and intensity of sheen and NAPL observations in native material decreased moving upstream of CM 3.1, and sheen and NAPL were not observed at any location between CM 3.2 and the head of the tributary.

In the Lower English Kills Category 2/3 Area, shake test sheen (Category 1A sheen) and positive NAPL shake test results (Category 1B and Category 2/3 NAPL) in the native material were observed from the sediment/native material interface and extending downward, generally 5 feet (south shoreline) to 12 feet (north shoreline) below the sediment/native material interface (corresponding to approximately 20 feet below the mudline). FS NAPL distribution refinement cores were collected and processed at 15 additional stations to refine the NAPL observations in native material in English Kills. As discussed in Section 5.4.3 of Appendix C, Category 2/3 NAPL was observed in five of the FS NAPL distribution refinement cores. The majority of NAPL observations in English Kills native material ranged from 1 to 24 cm (less than 1 to approximately 9 inches) thick and were generally located in sand beds in the native material. Cross-sections depicting visual observations and shake test results are presented in Figures C5-19 through C5-20m of Appendix C. Figure 4-142 shows the approximate lateral extent of Category 2/3 NAPL observed in native material. In the Lower English Kills Category 2/3 Area, Category 2/3 NAPL was present in a larger footprint in native material than in sediment. Where present in native material, Category 2/3 NAPL was observed at various depths and lengths in sand beds within the approximate upper 15 feet of the native material. The lateral extent was driven by 3- to 5-cm (less than 1- to 2-inch) observations of Category 2/3 NAPL in native material in the most upstream and most downstream cores.

4.6.5 Sediment Traps

4.6.5.1 Sediment Trap Dataset

As described in Sections 2.1.3.4 and 4.5.1, the Phase 2 sediment trap study was conducted at 30 locations in the Study Area for a 9-month period, starting in mid-June 2014 and extending through late March 2015 (see Figure 2-17a through d). As described in Section 4.5.1, the sediment traps collected depositing solids, and the traps were retrieved approximately once per month during the deployment period to document the physical characteristics of the material collected (as described in Section 4.3.1.2 of Appendix Bi), including visual observations of potential sheen and NAPL. Visual observations were recorded using Phase 2 terminology as described in Section 4.6.1.2; a shake test confirmation of visual observations was not performed. See Appendix Bi for more details on the sediment trap sampling program and sample collection methods. The results of the sediment trap gross solids deposition and chemical concentrations are summarized in Sections 4.5.2 and 4.5.3, respectively. The sediment trap samples consisted of generally fine-grained material (on average, 57 wt% to 89 wt%; see Section 4.5.2.2 and Table 4-35), with relatively high water content and low percent solids (on average, 13 wt% to 33 wt%; see Section 4.5.2.4 and Table 4-37). The distribution of visual observations of sheen and NAPL in the sediment traps, spatially and temporally, is discussed in Section 4.6.5.2.

4.6.5.2 Sheen and NAPL Observations

NAPL was not observed in any of the 239 Phase 2 sediment trap samples collected from the 30 sediment trap stations for physical characteristics; however, sheen was observed in 146 of the 239 sediment trap samples collected. Sheen was observed during at least one monitoring event at 29 of the 30 sediment trap stations; however, the frequency at which sheen was observed (i.e., the number of monthly monitoring events with sheen observations divided by the total number of monitoring events) at a given station varied widely. Phase 2 sediment trap visual observations of sheen and NAPL are summarized in Table 4-45 and Figure 4-143. As noted in Section 4.3.1.2 of Appendix Bi, not every sediment trap was monitored during each monthly monitoring event. In some cases, traps were not monitored for one or more of the following reasons: they moved or tipped over during the deployment period; they could not be located; or they could not be accessed due to weather conditions (e.g., the creek surface was frozen and the sampling station was inaccessible) or other safety related concerns.

Sheen was observed occasionally in CM 0 – 1 sediment traps, with the least frequent sheen observations at stations located near the mouth of the creek (see Figure 4-143). The one sediment trap station where sheen was not observed during any of the monitoring events (NC237ST) was located within this reach. Sheen observation frequency increased in CM 1 – 2 and in CM 2+ sediment traps. Sheen was frequently observed in sediment trap samples in

English Kills, with the greatest frequency observed at station EK004ST, a sediment trap station located in lower English Kills. On average, English Kills had the greatest frequency of sheen observations in sediment traps (see Figure 4-143). Fewer sheens were observed in East Branch, Maspeth Creek, and Whale Creek sediment traps, with sheen frequency in these tributaries being similar to the frequency of sheen observations in the CM 1 – 2 and CM 2+ sediment traps. In Dutch Kills, sheen frequency varied between the sediment trap stations in the lower and middle portions of the tributary compared with the station at the head of Dutch Kills, where sheen was observed more frequently. The sediment trap station at the head of Dutch Kills (DK011ST; see Figure 4-143) had the greatest frequency of sheen observations anywhere in the Study Area. High sheen frequencies (more than 78% of monitoring events) were also observed at stations NC111ST and EK004ST, which are located in CM 1 – 2 and lower English Kills, respectively (see Figure 4-143). Of the 30 stations monitored, the head of Dutch Kills was the only station where sheen was observed during every monitoring event.

Temporally, the number of sediment traps in which sheen was observed varied from month to month, as shown in Table 4-45. Sheen was observed at the fewest number of sampling stations during the January 2015 event, when sheen was observed in one-third (7 of 21) of the traps monitored. Sheen was observed at the greatest number of sampling stations during the March 2015 event, when sheen was observed in more than 90% (23 of 25) of the traps monitored. While the data indicate variability, there does not appear to be any consistent temporal differences in the number of sheen observations over the nine monitoring events.

Section 6.4.3.3 contains interpretation of the sediment trap data, including evaluation of potential factors that may have contributed to the observed temporal variability.

4.7 Surface Water

4.7.1 Surface Water Dataset

Surface water sampling and water quality profiling were conducted in the Study Area, Phase 1 reference areas, Phase 2 reference areas, and the East River to characterize surface water conditions to determine the nature and extent of impacts, as well as support the BERA and the BHHRA. Data were collected during both dry weather and wet weather conditions, as described in Section 2.1.4 and summarized as follows:

- Dry weather sampling was conducted during periodic (12 monthly) surveys in Phase 1 and twice during Phase 2 during conditions that generally did not capture effects from freshwater inflow during precipitation events.[114] There was no sampling strategy targeting a particular tide condition for the dry weather surface water data collection, although the overall collection effort resulted in samples being collected over the various stages of the tidal cycle (comparisons of the data by tide cycle are presented in this subsection). Phase 1 reference areas were sampled during a reconnaissance survey as part of the Phase 1 program, and the four Phase 2 reference areas were sampled in conjunction with the Phase 2 events.
- Wet weather sampling was conducted as part of the Phase 2 program to evaluate whether concentrations differed from those under the dry weather conditions sampled predominately during Phase 1. The wet weather program consisted of sampling throughout the Study Area (reference areas were not sampled), using the same sampling procedures as for dry weather, during five precipitation events of varying size between December 2014 and September 2015. Sampling was conducted concurrently with point source discharge sampling (see Section 5.1) and occurred on two consecutive days—the first during point source discharges (hereafter referred to as Round 1), and the second on the day after, which in most cases represented a condition in which the point source discharges had ceased or tailed off (hereafter referred to as Round 2). Results from Round 1 are compared to Round 2 in this subsection to evaluate the effects of transport, settling, and dilution within the Study Area. Also, the results from wet weather sampling are compared to the dry weather data in this subsection to evaluate the impacts associated with point source discharges on the system.

Water column samples were targeted to be collected from two depths: 1) a shallow sample generally collected within approximately 1 meter (approximately 3.3 feet) from the water surface (or in the case of Phase 2, from the mid-point depth of the upper stratified layer, in

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[114] Section 2.4.3.1 of the Phase 2 RI Work Plan Volume 1 included an assessment of conditions under which Phase 1 surface water data were collected (Anchor QEA 2014a). Most Phase 1 surface water sampling events were under dry weather, and for those that did include some sampling during or close to wet weather events, the salinity profile data showed only a thin layer of freshwater that was not represented by the 3-foot-deep shallow surface water samples.

cases where field measurements of salinity indicated stratified conditions); and 2) a deep sample collected within approximately 2 feet of the sediment surface.[115] Table 4-46 provides summary statistics for the dry weather sampling depths of shallow samples (in feet below the water surface) and deep samples (in feet above the sediment surface). The arithmetic average sampling depth of the shallow samples from the eight Study Area reaches ranged from 2.0 to 2.7 feet below the water surface. The arithmetic average depth of dry weather deep water samples from the eight Study Area reaches ranged from 0.7 to 3 feet above the sediment surface.[116] Table 4-47 provides summary statistics for the wet weather sampling depths. The arithmetic average shallow sampling depth ranged from 1.9 to 2.8 feet below the water surface for the eight Study Area reaches. The arithmetic average sampling depth of the deep samples ranged from 2.1 to 3.7 feet above the sediment surface for the eight Study Area reaches.[117]

A summary of data, for both dry and wet weather, is presented in the subsections that follow, first for certain conventional parameters, and then for TPAH (17), TPAH (34), TPCB, 2,3,7,8-TCDD, Cu, and Pb (surface water samples were not analyzed for C19-C36).[118]

4.7.2 Salinity, Organic Carbon, and TSS

Sections 4.7.2.1 through 4.7.2.3 provide an overview of spatial and temporal patterns in salinity, OC (particulate and dissolved), and TSS. These parameters provide several insights regarding freshwater inflows and tidal action, as well as distributions of chemical concentrations within the Study Area.

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[115] East River sampling, which is described more fully in Section 5.3, also included collection of an additional sample at near mid-depth in the water column.

[116] The reported depth for some dry weather deep samples was greater than the targeted 2 feet above the sediment surface. Due to either imprecision in water depth measurements or cases where sampling occurred during a falling tide, the sampling apparatus was positioned more than 2 feet above the sediment, so as to not come into contact with the sediment surface as the tide fell over the course of sample collection, which often took 1 to 2 hours per station, due to the large number of chemical analyses.

[117] Several wet weather deep samples were collected from a shallower depth than the targeted 2 to 3 feet above the sediment surface due to sampling equipment limitations in cases of water depths greater than 30 feet; however, the sampling depths in these cases were confirmed to be within the lower stratified layer based on observed salinity profiles.

[118] Two surface water samples were also collected as part of the gas ebullition pilot study conducted under Part 1 of the FS. Data from these samples are not included in this section, because they were only collected from two locations at one point in time and were only analyzed for a limited number of chemical constituents. The results from these two samples were within the range of the larger RI dataset, and the data are presented in Appendix Bii.

4.7.2.1 Salinity

Changes in salinity in surface water (temporally and/or spatially) provide insights regarding freshwater inputs and tidal action. Understanding where and when freshwater inputs relative to tidal flow occur can provide insight into the transport of chemicals in the Study Area and can be a potential indicator for different types of inflows, such as low salinity water from point source discharges and groundwater, compared to high salinity water from the East River.

Figure 4-144 shows the salinity measured during dry weather and wet weather sampling (surface and deep samples combined, with results shown for wet weather Rounds 1 and 2 separately) by reach. During dry weather, salinity is relatively consistent throughout the main stem, Dutch Kills, and Whale Creek, with values similar to those of the East River, while in Maspeth Creek, East Branch, and English Kills the salinity values are slightly lower (with the values in Maspeth Creek tending to be slightly lower than those in East Branch and English Kills). During wet weather, salinity is more variable, with more samples having decreased salinity relative to dry weather. This indicates an increased input of freshwater from point sources, overland flow, and precipitation into the Study Area during wet weather conditions. During the wet weather sampling, lower salinities were observed at several locations in the Round 1 samples (same day as discharges), as compared to Round 2 samples (day after discharges), as evidenced by the 25th percentile (bottom of the boxes in Figure 4-144) being generally lower in Round 1 compared to Round 2. Median salinity levels were marginally higher in Round 1 versus Round 2, in all reaches except CM 0 – 1.

With respect to depth, salinity in shallow samples was frequently lower during wet weather than during dry weather as shown in Figure 4-145. In contrast, salinity in the deeper samples was relatively consistent during dry and wet weather sampling events. The frequently lower salinity in the shallow samples indicates that, during wet weather, the freshwater influence from discharging point sources, overland flow, and precipitation is observed in the top portion of the water column. This is expected, due to the density differences between fresh (less dense) and saline (more dense) waters (additional discussion of this is provided in Section 6.2; see also Section 4.2.3 of Appendix G). Salinity in shallow water samples collected during wet weather was also more variable compared to the dry weather shallow samples and deep samples collected in dry and wet weather. This variability is likely a function of the timing between point source discharges and sample collection, as well as the potential presence of a relatively thin freshwater layer above the depth from which the shallow sample was collected. As such, the wet weather salinity was more variable in the Round 1 samples, as compared to Round 2 samples.

Figure 4-146 shows a time series of salinity measurements from samples collected during dry weather. Overall, salinity measurements during dry weather are relatively consistent, as would be expected. Infrequently, there are paired shallow and deep samples with results that vary from the remainder of the dataset at a given location, specifically the February results at all locations and the April results from CM 0 – 1. The cause of these anomalous results is unknown. In some cases, surface samples have lower salinity relative to the paired deeper samples; these likely reflect a small-scale freshwater influence, but these results are still generally much higher than values observed during the wet weather sampling.

Finally, variations of salinity with tide were evaluated by comparing dry weather data among different tidal conditions. Specifically, box plots of dry weather salinity by reach were developed comparing ebb versus flood tide[119] and by tide stage, which was separated into low tide, high tide, and median water.[120] Comparison of ebb and flood tide data allows for an evaluation of differences in flow direction and exchanges with the East River, whereas evaluating differences with tide height under dry weather may provide insight into the influence groundwater discharge may have on surface water. Figure 4-147a shows that salinity results in dry weather surface water collected during flood tide and ebb tide conditions were similar, indicating that exchange with the East River dominates dry weather conditions on a volumetric basis. Likewise, Figure 4-147b shows that salinity in dry weather surface water was generally similar among low tide, high tide, and median water conditions for all reaches. During high tide conditions, median salinity levels were marginally higher (i.e., by less than 1 or 2 psu in most cases) relative to low tide or median water conditions, for six of the eight reaches (which could be a result of inflow from fresh groundwater).

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[119] Results in which sample collection started during one tide phase and ended in the other were excluded from this analysis. A sample was defined as being sampled during ebb or flood tide conditions if the start and end time for sample collection occurred before low or high tide, respectively.

[120] Tide height conditions were defined using surface water elevation data collected from the National Grid tidal gauge. Using these data, the surface water samples were assigned to tide heights using the following criteria: low tide (0.5 foot less than the annual median water surface elevation); high tide (0.5 foot greater than the annual median water surface elevation); and median water (within +/- 0.5 foot of the annual median water surface elevation).

Regardless of the tidal condition, salinity measurements were marginally lower in the upstream tributaries relative to the main stem.

4.7.2.2 Organic Carbon

Evaluation of particulate and dissolved phase OC concentrations in surface water samples is important to understand the partitioning of chemicals between the dissolved and particulate phases in surface water samples (see Section 6.4.1) and distributions of chemical concentrations in the water column.

Figure 4-148 shows the particulate organic carbon (POC) concentrations in surface water during dry and wet weather (surface and deep samples combined). During dry weather, POC concentrations increased somewhat from the mouth of Newtown Creek, up through the upper tributaries, and were generally similar to those in the Phase 2 reference areas. Similarly, wet weather POC concentrations show an increase with distance upstream, and are highest in the upper tributaries. Further, wet weather POC concentrations are greater than those observed during dry weather throughout the Study Area. Average POC concentrations measured in Round 2 generally tended to be lower than those measured in Round 1, and the Round 2 concentrations often overlapped with the dry weather interquartile ranges.

A common way of evaluating POC in surface water is to normalize it to the TSS concentration, which results in the fraction organic carbon (f_{OC}) of the suspended solids (with units of wt% on a dry weight basis). The results for surface water f_{OC} are shown in Figure 4-149 (surface and deep samples combined). During dry weather, f_{OC} increases with distance upstream from a median of approximately 5 wt% at the mouth to approximately 10 wt% in the upstream tributaries. The median dry weather f_{OC} from the lower Study Area (CM 0 – 1) is generally similar to the median values of the Phase 2 reference areas; although median values increase in the Study Area going upstream, the majority of f_{OC} values in these reaches (as represented by the interquartile range boxes in Figure 4-149) are generally within those of the Spring Creek reference area (as well as the Head of Bay reference area when maximum values are considered). The increased dry weather f_{OC} in the upper portion of the Study Area may also be related in part to a contribution of suspended solids from algal production (see Section 6.3 for more discussion). During wet weather, f_{OC} increases relative to dry weather, to median values in the range of 10 wt% to 20 wt%. This increase during wet weather is expected, because of point source loads, particularly the higher organic inputs from CSOs (see Section 5.1.3.3.2, which indicates POC of point sources data averages 14 wt% or higher). Figure 4-150 shows the time series of surface water f_{OC} data within the Study Area. Although some of the highest f_{OC} values were measured during the warmer summer months (also consistent with some contribution from internal algal production), the trend is not consistent by reach, and the f_{OC} is generally variable throughout the year.

Nearly all the dry weather dissolved organic carbon (DOC) data collected during Phase 1 and part of Phase 2 were reported as non-detect (at detection limits that ranged from 1 to 20 mg/L), due to relatively low concentrations and laboratory detection limit issues. However, DOC samples collected during the last four wet weather events from Phase 2 were analyzed by a different laboratory, resulting in DOC detections in 146 of 154 samples. DOC results for these wet weather samples are relatively low, ranging from 2 to 4 mg/L, and show no spatial gradient (see Figure 4-151). These low concentrations are similar to other New York Harbor datasets, including the CARP and NYCDEP Harbor sampling data from the East River and NYCDEP Harbor data from the Study Area (see Table 2-1). The NYCDEP Harbor sampling data for the station in the East River (Station E2, near the mouth of Newtown Creek) have DOC concentrations ranging from 1.4 to 5.9 mg/L, with an arithmetic average of 3.2 mg/L; the relatively limited number of samples from the CARP dataset in the East River have DOC concentrations ranging from 2.6 to 21 mg/L, with an arithmetic average of 7.1 mg/L and a median of 3.6 mg/L. The NYCDEP Harbor sampling data from within the Study Area have DOC concentrations ranging from 2.2 to 9.0 mg/L, with an arithmetic average of 3.6 mg/L. Table 4-48 shows summary statistics for these NYCDEP Harbor and CARP program DOC results (NYCDEP 2018b; NYSDEC 2018b).

4.7.2.3 TSS

TSS concentrations measured during dry and wet weather are shown in Figure 4-152 (surface and deep samples combined). During dry weather, TSS concentrations vary by a factor of up to 10 at a given location, but the central tendencies are relatively consistent on a spatial basis throughout the Study Area, with median values in the range of 15 to 30 mg/L. Dry weather TSS concentrations in the Study Area are also consistent with concentrations measured in both the Phase 2 reference areas and the East River.

During wet weather, TSS concentration ranges are similar to those observed during dry weather. Although increases in TSS concentration may be expected due to solids loading from point source discharges during wet weather, dilution from increased water inflow would act to offset those increases to some extent. Furthermore, it should be recognized that differences between the wet and dry weather TSS data are difficult to discern, due to the following factors: 1) the relatively large variability in dry and wet weather TSS concentrations; 2) the relatively small difference between TSS concentrations in Study Area surface water and point source discharges (i.e., point source TSS concentrations generally average around 100 mg/L; see Section 5.1.3.3.1)[121]; and 3) the fact that the wet weather sampling program design could not include sampling of ambient conditions prior to the onset of the precipitation event. Nonetheless, concentrations during wet weather were generally higher for Round 1 relative to Round 2, which is likely indicative of solids inputs from point source discharges. This decrease in TSS concentrations measured during Round 2 is likely due to mixing within the water column and settling losses that would be expected when new solids enter the system via point source discharges or overland flow.

The distributions of TSS concentrations by Study Area reach during both dry and wet weather generally do not vary appreciably with depth in the water column, although Round 1 to Round 2 wet weather differences are observed in both shallow and deep sample data (see Figure 4-153). To evaluate seasonal variation, Figure 4-154 shows time series of dry weather TSS concentrations measured each month in the various reaches of the Study Area. This figure shows that dry weather TSS concentrations are variable and generally do not show any distinct seasonal pattern, although there is some tendency for higher values during summer months in some reaches, which may be a reflection of some contribution from internal algal production (see Sections 4.7.2.2 and 6.3).

Additional evaluations of TSS were performed as part of the sediment transport modeling (see Section 6.3; see also Section 5.2.4 of Appendix G), including a sampling program that characterized the grain size distribution of solids in samples collected from the East River and Study Area for Part 1 of the FS.

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[121] In contrast, differences between chemical concentrations for point source samples and dry weather surface water samples are much greater (often an order of magnitude), which provides for a much stronger signal to evaluate point source influences, as discussed in Section 4.7.4.

4.7.3 Dry Weather Concentrations

4.7.3.1 TPAH (17)

4.7.3.1.1 Spatial Distribution

TPAH (17) concentrations in surface water measured during dry weather sampling events from each reach of the Study Area and the four Phase 2 reference areas are summarized in Table 4-49; a box plot showing the data by reach, including data from the East River (discussed further in Section 5.3), is provided in Figure 4-155.[122]

Overall, TPAH (17) concentrations increase with distance upstream through the main stem and into the upstream tributaries. The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1, CM 1 – 2, and CM 2+**. TPAH (17) concentrations show a slight increasing gradient in the main stem with distance upstream. However, concentrations in all three of these main stem reaches are generally within the same range as the East River samples, as well as the Westchester Creek and Spring Creek Phase 2 reference areas. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **CM 0 – 1**. TPAH (17) concentrations range from 0.0023 to 0.66 µg/L (arithmetic average = 0.13 µg/L; median = 0.11 µg/L).
  - **CM 1 – 2**. TPAH (17) concentrations range from 0.0021 to 0.49 µg/L (arithmetic average = 0.15 µg/L; median = 0.13 µg/L).
  - **CM 2+**. TPAH (17) concentrations range from 0.0019 to 0.83 µg/L (arithmetic average = 0.17 µg/L; median = 0.14 µg/L).
- **English Kills and East Branch**. The highest TPAH (17) concentrations are observed in samples collected from these upstream tributaries, particularly English Kills. Specifically, the arithmetic average dry weather TPAH (17) concentration in English Kills is approximately 2 times higher than in CM 0 – 1 and CM 1 – 2 (median concentrations are between 1.5 to 1.7 times greater in English Kills than in CM 0 – 1

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[122] The evaluations of surface water spatial patterns provided throughout this section combine shallow and deep samples, unless otherwise noted.

and CM 1 – 2). Concentrations are also higher compared to the four Phase 2 reference areas. Ranges, arithmetic averages, and medians of concentrations are as follows:

  - **English Kills**. TPAH (17) concentrations range from 0.064 to 1.2 µg/L (arithmetic average = 0.27 µg/L; median = 0.19 µg/L).
  - **East Branch**. TPAH (17) concentrations range from 0.081 to 0.55 µg/L (arithmetic average = 0.20 µg/L; median = 0.17 µg/L).

- **Maspeth Creek, Dutch Kills, and Whale Creek**. TPAH (17) concentrations in these three tributaries are generally similar to one another, and are consistent with those observed in the main stem, the East River, and the Westchester Creek and Spring Creek Phase 2 reference areas. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **Maspeth Creek**. TPAH (17) concentrations range from 0.071 to 0.29 µg/L (arithmetic average = 0.16 µg/L; median = 0.15 µg/L).
  - **Dutch Kills**. TPAH (17) concentrations range from 0.0013 to 0.52 µg/L (arithmetic average = 0.16 µg/L; median = 0.14 µg/L).
  - **Whale Creek**. TPAH (17) concentrations range from 0.054 to 0.39 µg/L (arithmetic average = 0.16 µg/L; median = 0.16 µg/L).

4.7.3.1.2 Variations with Depth, Time, and Tidal Cycle

To evaluate possible differences with depth in the water column, the concentration of TPAH (17) in paired shallow and deep samples from the Study Area and Phase 2 reference areas collected during dry weather are compared (using cross plots) in Figure 4-156. In the main stem and in the Phase 2 reference areas, concentrations of TPAH (17) in surface and deep samples are variable, but did not show a systematic difference with sampling depth. In the tributaries, some differences are observed in Figure 4-156 at the higher end of the concentration range. Specifically, for concentrations less than 0.5 µg/L in the tributaries, there is no apparent systematic difference between surface samples and deep samples (i.e., the paired data scatter on either side of the 1:1 line in the middle panel of Figure 4-156). In contrast, for concentrations greater than 0.5 µg/L in the tributaries, the concentration of the deep sample is greater than that of the corresponding shallow sample (i.e., symbols below the 1:1 line) for six of seven samples, most of which are from English Kills.

Time series of dry weather surface water TPAH (17) concentrations are shown in Figure 4-157. This figure shows that TPAH (17) concentrations are somewhat variable during the year, but do not appear to show any clear pattern with season. One exception is that the highest concentrations tended to be observed in some samples collected during summer months in some reaches (e.g., English Kills and Dutch Kills). However, these differences are generally not large (i.e., factor of 2 or 3).

Figure 4-158a compares TPAH (17) concentrations by Study Area reach, based on the tidal flow direction during which the sample was collected. Concentrations in surface water collected during flood tide and ebb tide were similar, suggesting that within the timescale of a tidal cycle, tidal currents have little influence on the concentration of TPAH (17) in Study Area surface water. Figure 4-158b compares TPAH (17) concentrations by reach, based on the tide stage during which the sample was collected. Most reaches show similar concentrations among low tide, high tide, and median water conditions. Although there is variability, some reaches, like Whale Creek and English Kills, show somewhat lower concentrations at low tide compared to high tide and median water.

4.7.3.2 TPAH (34)

4.7.3.2.1 Spatial Distribution

Concentrations of TPAH (34) in surface water measured during dry weather sampling events from each reach of the Study Area and the four Phase 2 reference areas are summarized in Table 4-50; a box plot showing the data by reach, and the East River (discussed further in Section 5.3), is provided in Figure 4-159.

Overall, TPAH (34) concentrations increase with distance upstream through the main stem and into the upstream tributaries. The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1, CM 1 – 2, and CM 2+**. TPAH (34) concentrations show an increasing longitudinal gradient in the main stem with distance upstream. This gradient is somewhat greater than that of TPAH (17) (compare Figure 4-159 to 4-155). Concentrations of TPAH (34) in CM 0 – 1 are in the same general range as those in the

East River and the Westchester Creek reference area, and generally higher in CM 1 – 2 and CM 2+. Ranges, arithmetic averages, and medians of concentrations are as follows:

  - **CM 0 – 1**. TPAH (34) concentrations range from 0.059 to 2.9 μg/L (arithmetic average = 0.33 μg/L; median = 0.21 μg/L).
  - **CM 1 – 2**. TPAH (34) concentrations range from 0.046 to 1.6 μg/L (arithmetic average = 0.43 μg/L; median = 0.36 μg/L).
  - **CM 2+**. TPAH (34) concentrations range from 0.075 to 2.6 μg/L (arithmetic average = 0.53 μg/L; median = 0.46 μg/L).

- **Tributaries**. The highest TPAH (34) concentrations are observed in samples collected from the upstream tributaries (i.e., English Kills, East Branch, and Maspeth Creek, where arithmetic averages are 1.7 to 2.2 times greater than in CM 0 – 1). TPAH (34) arithmetic average values in Whale Creek and Dutch Kills are within the range of CM 0 – 2 values. Arithmetic average concentrations in the Study Area tributaries are also higher than those in the four Phase 2 reference areas (although the interquartile ranges for Whale Creek and Maspeth Creek are similar to that of the Westchester Creek reference area). Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **English Kills**. TPAH (34) concentrations range from 0.072 to 3.1 μg/L (arithmetic average = 0.73 μg/L; median = 0.63 μg/L).
  - **East Branch**. TPAH (34) concentrations range from 0.15 to 1.1 μg/L (arithmetic average = 0.61 μg/L; median = 0.62 μg/L).
  - **Maspeth Creek**. TPAH (34) concentrations range from 0.12 to 1.4 μg/L (arithmetic average = 0.57 μg/L; median = 0.48 μg/L).
  - **Dutch Kills**. TPAH (34) concentrations range from 0.056 to 1.1 μg/L (arithmetic average = 0.36 μg/L; median = 0.32 μg/L).
  - **Whale Creek**. TPAH (34) concentrations range from 0.089 to 1.6 μg/L (arithmetic average = 0.42 μg/L; median = 0.40 μg/L).

4.7.3.2.2 Variations with Depth, Time, and Tidal Cycle

To evaluate possible differences with depth in the water column, the concentration of TPAH (34) in paired shallow and deep samples from the Study Area and Phase 2 reference areas collected during dry weather are compared (using cross plots) in Figure 4-160. The Study Area main stem and tributaries showed somewhat higher TPAH (34) concentrations reported for the shallow sample depth as compared to the deep sample depth (binomial test p-values of <0.05), although there is a large amount of variability. In the Reference Areas, deep samples tended to exhibit higher concentrations than shallow samples (11 of 15 samples are below the 1:1 line, with a binomial test p-value of 0.059), although the magnitude of concentration differences is small.

Time series of dry weather surface water TPAH (34) concentrations are shown in Figure 4-161. This figure shows that TPAH (34) concentrations are somewhat variable during the year but do not appear to show any clear and consistent seasonal patterns. There are some sampling events during which a singularly high concentration was observed (e.g., Whale Creek in November or CM 2+ in February, March, and November). There are also instances where several samples and depths from a single sampling month are elevated relative to the other months (e.g., English Kills in September and December). However, overall, there are no discernable seasonal patterns, other than possibly at Dutch Kills, where the data suggest somewhat higher concentrations in the summer months as compared to other times of the year.

Figure 4-162a compares TPAH (34) concentrations by Study Area reach, based on the tidal flow direction during which the sample was collected. Concentrations in surface water collected during flood tide and ebb tide were generally similar in each of the Study Area reaches, with the possible exception of CM 0 – 1, where concentrations during flood tide were less variable and somewhat lower than at ebb tide. Overall, the general similarity suggests that within the timescale of a tidal cycle, tidal currents have little influence on the concentration of TPAH (34) in Study Area surface water. Figure 4-162b compares TPAH (34) concentrations by reach, based on the tide stage during which the sample was collected. Most reaches show similar concentrations among low tide, high tide, and median water conditions. One exception appears to be English Kills, which shows somewhat lower concentrations at low tide compared to high tide and median water.

4.7.3.3 TPCB

All Phase 1 surface water samples were analyzed using an Aroclor-based laboratory method (Method 8082A); approximately 25% of these Phase 1 samples were also analyzed using a higher resolution congener-specific method (Method 1668). All Phase 2 samples were analyzed using Method 1668. Most of the samples analyzed for PCB congeners have detected TPCB results (97%), so the congener PCB dataset was the primary dataset used to evaluate nature and extent of surface water PCBs within the Study Area. In contrast, only 16% of the lower resolution Phase 1 Aroclor samples have detected results for TPCB. Therefore, PCB Aroclor results are only discussed qualitatively in the evaluations described in this section.

4.7.3.3.1 Spatial Distribution

TPCB congener concentrations in surface water measured during dry weather sampling events from each reach of the Study Area and the four Phase 2 reference areas are summarized in Table 4-51, and a box plot showing the data by reach, including data from the East River (discussed further in Section 5.3), is provided in Figure 4-163. Arithmetic average and median concentrations within the Study Area reaches exceed those in the Phase 2 reference areas. For comparison, Figure 4-164 is a longitudinal profile of individual Phase 1 surface water TPCB Aroclor concentrations (a box plot was not created using TPCB Aroclor results due to the low FoD). Summary statistics for Study Area and Phase 1 reference area TPCB Aroclor concentrations are provided in Table 4-52.

The overall trend for TPCB congener concentrations in Newtown Creek during dry weather conditions is an increasing concentration with increasing distance upstream, including into the upstream tributaries. The range, arithmetic average, and median (TPCB congener concentrations) for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1, CM 1 – 2, and CM 2+**. TPCB congener concentrations in the main stem show little spatial pattern, although concentrations in CM 2+ are somewhat higher than those in the other two more downstream reaches. The highest concentration observed in the Study Area during dry weather was in CM 2+ (92 nanograms per liter [ng/L]). TPCB congener concentrations in these reaches are somewhat higher than those observed at the East River locations, although the distributions of the data generally overlap. Lastly, main stem TPCB congener concentrations are generally higher than the four Phase 2 reference areas. Ranges, arithmetic averages, and medians of concentrations are as follows:

  - **CM 0 – 1**. TPCB concentrations range from 2.4 to 12 ng/L (arithmetic average = 6.8 ng/L; median = 6.7 ng/L).
  - **CM 1 – 2**. TPCB concentrations range from 0.66 to 18 ng/L (arithmetic average = 6.6 ng/L; median = 6.3 ng/L).
  - **CM 2+**. TPCB concentrations range from 2.7 to 92 ng/L (arithmetic average = 9.5 ng/L; median = 7.1 ng/L).

- **English Kills**. TPCB concentrations range from 16 to 18 ng/L (arithmetic average and median = 17 ng/L). The highest dry weather TPCB congener concentrations are observed in samples collected from this tributary. Specifically, the arithmetic average and median concentrations in English Kills are nearly 2 to 3 times higher than that in the main stem and other tributaries. However, it should be noted that comparison of tributary TPCB data with data collected in the main stem is confounded by low sample counts in the tributaries (only two to four samples per tributary). TPCB concentrations in English Kills are also greater than those in all reference area samples.
- **East Branch, Maspeth Creek, and Dutch Kills**.[123] TPCB congener concentrations in these three tributaries are generally consistent with those observed in the nearby main stem. Similar to the main stem concentrations, TPCB levels in these tributaries are slightly higher than those collected in the East River, although by generally less than a factor of 2 (recognizing the lower sample counts in the tributaries as noted in the previous bullet). TPCB concentrations from these three tributaries are greater than those from the reference areas, for all but one sample. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **East Branch**. TPCB concentrations range from 5.5 to 11 ng/L (arithmetic average and median = 8.0 ng/L).
  - **Maspeth Creek**. TPCB concentrations range from 3.1 to 9.1 ng/L (arithmetic average = 7.0 ng/L; median = 8.7 ng/L).
  - **Dutch Kills**. TPCB concentrations range from 5.8 to 6.4 ng/L (arithmetic average = 6.1 ng/L; median = 6.0 ng/L).

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[123] Sampling for TPCB in Whale Creek was only conducted during Phase 1, using the Aroclor-based method.

Despite the low FoD in the Phase 1 TPCB Aroclor dataset, the data are qualitatively consistent with the spatial patterns evident in the TPCB congener data described previously (see Figure 4-164 for Aroclor data).

4.7.3.3.2 Variations with Depth, Time, and Tidal Cycle

To evaluate possible differences with depth in the water column, the concentration of TPCB congeners in paired shallow and deep samples from the Study Area and Phase 2 reference areas collected during dry weather are compared (using cross plots) in Figure 4-165. This figure shows there is no clear relationship between dry weather TPCB concentration and depth. There are limited paired shallow and deep samples from the tributaries, because no PCB congener samples were collected in Phase 1, and a shallow water column at most locations prevented sample collection at two depths. In the reference areas, deep samples exhibit higher concentrations than shallow samples (eight of nine samples are below the 1:1 line, with a binomial test p value of 0.020), although the magnitude of concentration differences is small.

Figure 4-166 shows time series of dry weather TPCB congener concentrations for different reaches. This figure shows that there is evidence of a seasonal pattern in TPCB concentrations within the main stem of Newtown Creek. TPCB concentrations in CM 0 – 1 and CM 1 – 2 are higher by a factor of approximately 3 to 4 during the warmer months, relative to the colder winter months. This pattern is less evident for the data in CM 2+, which may be attributable to differences in the influence of the East River (i.e., inflows from the East River have a smaller impact further upstream at CM 2+), which also exhibits a similar seasonality (see Section 5.3.4 for more discussion of this topic). The temporal patterns of TPCB concentrations are different than those observed for TPAH (17) and TPAH (34) (see Sections 4.7.3.1.2 and 4.7.3.2.2, respectively); specifically, the temporal pattern observed in the main stem for TPCB is weaker, or not observed, for TPAH (17) and TPAH (34). Additional interpretive discussions related to chemical sources are provided in Section 6.4.

There is no clear relationship between TPCB concentration and phase of the tidal cycle during which samples were collected based on both tidal flow direction (i.e., ebb versus flood; see Figure 4-167a) and tide stage (i.e., low tide, high tide, median water; see Figure 4-167b).

4.7.3.4 2,3,7,8-TCDD

2,3,7,8-TCDD concentrations in surface water measured during dry weather sampling events from each reach of the Study Area and the four Phase 2 reference areas are summarized in Table 4-53; a box plot showing the data, and the East River, by reach is provided in Figure 4-168.

Out of 154 dry weather samples, including samples collected from the Phase 2 reference areas, only one sample from CM 1 – 2 had a detectable 2,3,7,8-TCDD concentration. Therefore, there are no spatial or temporal patterns of 2,3,7,8-TCDD in surface water under dry weather conditions within the limits of detection.

4.7.3.5 Cu

4.7.3.5.1 Spatial Distribution

Concentrations of Cu in surface water measured during dry weather sampling events from each reach of the Study Area and the four Phase 2 reference areas are summarized in Table 4-54; a box plot showing the data by reach, and the East River (discussed further in Section 5.3), is provided in Figure 4-169.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1, CM 1 – 2, and CM 2+**. There is no apparent spatial pattern in Cu within the main stem reaches. However, several higher concentration samples are found within CM 2+, which result in a somewhat higher arithmetic average concentration in this reach (but not an elevated median concentration). Dry weather Cu concentrations within the main stem are generally somewhat lower than the concentrations in samples from the East River, although the range of concentrations generally overlaps. Lastly, Cu concentrations in these three reaches overlap those observed in the four Phase 2 reference areas, with the exception of a few high concentration samples in each reach. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **CM 0 – 1**. Cu concentrations range from 1.1 to 17 µg/L (arithmetic average = 4.3 µg/L; median = 4.0 µg/L).

  - **CM 1 – 2**. Cu concentrations range from 0.1 to 13 µg/L (arithmetic average = 4.4 µg/L; median = 4.7 µg/L).
  - **CM 2+**. Cu concentrations range from 1.0 to 90 µg/L (arithmetic average = 6.2 µg/L; median = 4.2 µg/L).
- **Tributaries**. Dry weather Cu concentrations are relatively similar across all the tributaries, with the exception of a slightly higher arithmetic average in Whale Creek (however, the median concentration in Whale Creek is similar to the other tributaries). Concentrations in the tributaries are also generally consistent with those observed in the main stem and in the Phase 2 reference areas. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **English Kills**. Cu concentrations range from 1.0 to 12 µg/L (arithmetic average = 3.5 µg/L; median = 3.2 µg/L).
  - **East Branch**. Cu concentrations range from 1.0 to 23 µg/L (arithmetic average = 4.5 µg/L; median = 3.2 µg/L).
  - **Maspeth Creek**. Cu concentrations range from 2.0 to 5.1 µg/L (arithmetic average = 3.8 µg/L; median = 4.0 µg/L).
  - **Dutch Kills**. Cu concentrations range from 0.1 to 22 µg/L (arithmetic average = 3.5 µg/L; median = 2.8 µg/L).
  - **Whale Creek**. Cu concentrations range from 1.9 to 17 µg/L (arithmetic average = 5.0 µg/L; median = 3.7 µg/L).

4.7.3.5.2 Variations with Depth, Time, and Tidal Cycle

To evaluate possible differences with depth in the water column, the concentration of Cu in paired shallow and deep samples from the Study Area and Phase 2 reference areas collected during dry weather are compared (using cross plots) in Figure 4-170. Higher concentrations of Cu in the main stem locations were measured in the deeper samples in a majority of paired samples (binomial test p value of 0.026). When both the shallow and deep paired samples have relatively low concentrations (e.g., 5 µg/L or less), there is no clear relationship between sampling depth and Cu concentration. Cu concentrations are not dependent on depth in the tributaries and reference areas.

In addition, Cu concentrations measured during dry weather, although variable, do not appear to be dependent on seasonal changes (see Figure 4-171) or tide cycle (see Figures 4-172a and 4-172b).

4.7.3.6 Pb

4.7.3.6.1 Spatial Distribution

Concentrations of Pb in surface water measured during dry weather sampling events from each reach of the Study Area and the four Phase 2 reference areas are summarized in Table 4-55; a box plot showing the data by reach, and the East River, is provided in Figure 4-173.

Most dry weather surface water Pb samples are less than the method detection limits (MDLs) (21% FoD across Study Area samples; see Table 4-55). As a result, spatial and temporal patterns are difficult to discern. When comparing the results between Phase 1 and Phase 2 sampling, it was observed that many of the non-detects were a result of samples from the Phase 1 program having a higher detection limit (10 µg/L), with most of the detected results coming from samples with a lower detection limit. Using only the Phase 2 data, which had a lower detection limit (maximum MDL of 4 µg/L), the FoD of samples collected in the Study Area was higher, at 52%. Figure 4-174 shows the concentrations of Pb measured in the dry weather program for only Phase 2 data, so the effects of the lower FoD in Phase 1 (i.e., by comparing to Figure 4-173) can be evaluated. In general, both treatments of the data show a lack of spatial pattern.

The range, arithmetic average, and median for each reach and notable patterns in the data (based on both Phase 1 and Phase 2) are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1, CM 1 – 2, and CM 2+**. There is no apparent spatial pattern in Pb within the main stem reaches. Dry weather Pb concentrations within the main stem are generally slightly lower than the concentrations in samples from the East River, although the range of concentrations generally overlaps. Pb concentrations in these three reaches overlap those observed in the four Phase 2 reference areas. Ranges, arithmetic averages, and medians of concentrations are as follows:

  - **CM 0 – 1**. Pb concentrations range from 0.7 to 10 µg/L (arithmetic average = 4.0 µg/L; median = 2.0 µg/L).
  - **CM 1 – 2**. Pb concentrations range from 0.5 to 10 µg/L (arithmetic average = 4.9 µg/L; median = 2.4 µg/L).
  - **CM 2+**. Pb concentrations range from 0.6 to 16 µg/L (arithmetic average = 4.8 µg/L; median = 2.0 µg/L).

- **Tributaries**. Dry weather Pb concentrations are relatively similar across all the tributaries for both Phase 1 and Phase 2 samples. Concentrations in the tributaries are also generally consistent with those observed in the main stem and in the Phase 2 reference areas. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **English Kills**. Pb concentrations range from 1.5 to 10 µg/L (arithmetic average = 4.2 µg/L; median = 2.0 µg/L).
  - **East Branch**. Pb concentrations range from 1.7 to 10 µg/L (arithmetic average = 4.6 µg/L; median = 2.0 µg/L).
  - **Maspeth Creek**. Pb concentrations range from 1.7 to 10 µg/L (arithmetic average = 4.5 µg/L; median = 2.0 µg/L).
  - **Dutch Kills**. Pb concentrations range from 0.5 to 10 µg/L (arithmetic average = 4.2 µg/L; median = 2.0 µg/L).
  - **Whale Creek**. Pb concentrations range from 1.7 to 15 µg/L (arithmetic average = 4.7 µg/L; median = 2.0 µg/L).

4.7.3.6.2 Variations with Depth, Time, and Tidal Cycle

To evaluate possible differences with depth in the water column, the concentration of Pb in paired shallow and deep samples from the Study Area and Phase 2 reference areas collected during dry weather are compared (using cross plots) in Figure 4-175. Higher concentrations of Pb in the main stem locations were measured in the deeper samples in a majority of paired samples (binomial test p value of 0.073), although most of the results were below the detection limit and not included in this figure. Pb concentrations are not dependent on depth in the tributaries. In reference areas samples, the shallow samples often had concentrations greater than the deeper samples, although the results were not statistically significant (binomial test p value of 0.14).

In addition, Pb concentrations measured during dry weather do not appear to be dependent on seasonal changes (see Figure 4-176) or tide cycle (see Figures 4-177a and 4-177b), although the low FoD limits the ability to evaluate patterns in the data.

4.7.4 Wet Weather Concentrations

4.7.4.1 TPAH (17)

4.7.4.1.1 Spatial Distribution

TPAH (17) concentrations in surface water measured during wet weather sampling events from each reach of the Study Area are summarized in Table 4-56; a box plot showing the wet weather data (compared to dry weather) by reach is provided in Figure 4-178. Note that the statistics shown in Table 4-56 combine results from Round 1 and Round 2 wet weather sampling (as well as shallow and deep depths); Figure 4-178 shows the Round 1 and Round 2 sampling events separately.[124]

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1, CM 1 – 2, and CM 2+**. Wet weather TPAH (17) concentrations show evidence of an increase with distance upstream in the main stem. Specifically, concentrations are lowest in CM 0 – 1 (nearest the East River) and then increase by a factor of 1.5 to 2 in CM 1 – 2 and CM 2+. This pattern suggests an increased effect from point sources with distance upstream, and less influence from the East River. Also, arithmetic average wet weather TPAH (17) concentrations in these main stem areas range from 2 to 3 times higher than the corresponding dry weather concentrations. These observed increases during wet weather are also suggestive of an influence from point source discharges. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **CM 0 – 1**. TPAH (17) concentrations range from 0.10 to 0.82 µg/L (arithmetic average = 0.25 µg/L; median = 0.21 µg/L).
  - **CM 1 – 2**. TPAH (17) concentrations range from 0.11 to 2.2 µg/L (arithmetic average = 0.44 µg/L; median = 0.26 µg/L).

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[124] This same data treatment is used for the presentation of TPAH (34), TPCB, 2,3,7,8-TCDD, Cu, and Pb in the remainder of this section.

  - **CM 2+**. TPAH (17) concentrations range from 0.10 to 1.2 µg/L (arithmetic average = 0.41 µg/L; median = 0.30 µg/L).

- **Tributaries.**[125] Similar to dry weather conditions, the highest wet weather TPAH (17) concentrations are generally observed in the tributaries. Specifically, arithmetic average TPAH (17) concentrations in the tributaries are 1.5 to 2 times higher than those observed in CM 1 – 2 and CM 2+, and up to 3 times higher than those in CM 0 – 1. Also, arithmetic average wet weather TPAH (17) concentrations in the tributaries range from 2 to 5 times higher than the corresponding dry weather concentrations. This larger range in observed increases during wet weather (relative to the main stem areas) suggests a larger influence of point source discharges within the tributaries. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **English Kills**. TPAH (17) concentrations range from 0.096 to 3.3 µg/L (arithmetic average = 0.62 µg/L; median = 0.41 µg/L).
  - **East Branch**. TPAH (17) concentrations range from 0.078 to 1.7 µg/L (arithmetic average = 0.60 µg/L; median = 0.45 µg/L).
  - **Maspeth Creek**. TPAH (17) concentrations range from 0.17 to 2.5 µg/L (arithmetic average = 0.74 µg/L; median = 0.47 µg/L).
  - **Dutch Kills**. TPAH (17) concentrations range from 0.16 to 4.1 µg/L (arithmetic average = 0.75 µg/L; median = 0.40 µg/L).

4.7.4.1.2 Comparison Between Round 1 and Round 2 Sampling

Figure 4-178 shows evidence that TPAH (17) concentrations in samples collected in Round 1 were greater than the samples collected in Round 2. This result is expected, due to the mixing and dilution that occur in surface water during and following the cessation of the point source discharges. Figure 4-179 further evaluates differences between Round 1 and Round 2 sampling at all locations, by comparing TPAH (17) concentrations measured in Round 1 versus Round 2 (cross plots) for each individual wet weather sampling event. Round 1 concentrations were more often greater than Round 2, as indicated by p values < 0.05 from binomial tests for three of the five events. In most events, when TPAH (17)

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[125] Whale Creek was not included in the Phase 2 wet weather sampling program.

concentrations were elevated, the Round 1 shallow (near-surface) sample concentrations were often greater than the Round 2 shallow samples (denoted as circles in Figure 4-179). Deep sample results (denoted as squares in Figure 4-179) were often clustered more closely around the 1:1 line, indicating that the deep sample concentrations from Rounds 1 and 2 were similar. This pattern indicates a decrease in near-surface concentrations between the time of discharge and the next day, which is suggestive of mixing and deposition processes. This pattern was not observed for all five sampling events, indicating variability associated with amount, intensity, and timing of precipitation.

4.7.4.2 TPAH (34)

4.7.4.2.1 Spatial Distribution

Concentrations of TPAH (34) in surface water measured during wet weather sampling events from each reach of the Study Area are summarized in Table 4-57; a box plot showing the wet weather data (compared to dry weather) by reach is provided in Figure 4-180.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1, CM 1 – 2, and CM 2+**. Similar to the patterns observed in TPAH (17), TPAH (34) concentrations show evidence of an increase with distance upstream in these main stem reaches, with the lowest concentrations in the area closest to the East River (CM 0 – 1). Also, arithmetic average wet weather TPAH (34) concentrations in these main stem areas range from 2 to 3 times higher than the corresponding dry weather concentrations. These observed increases during wet weather indicate an influence from point source discharges. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **CM 0 – 1**. TPAH (34) concentrations range from 0.16 to 3.4 µg/L (arithmetic average = 0.77 µg/L; median = 0.54 µg/L).
  - **CM 1 – 2**. TPAH (34) concentrations range from 0.26 to 6.1 µg/L (arithmetic average = 1.3 µg/L; median = 0.69 µg/L).
  - **CM 2+**. TPAH (34) concentrations range from 0.13 to 3.6 µg/L (arithmetic average = 1.3 µg/L; median = 0.95 µg/L).

- **Tributaries**. Arithmetic average TPAH (34) concentrations in the tributaries are 1.3 to 2 times higher than those observed in CM 1 – 2 and CM 2+ and up to 3.4 times higher than those in CM 0 – 1. Also, arithmetic average wet weather TPAH (34) concentrations in the tributaries range from 2.3 to 6.1 times higher than the corresponding dry weather concentrations. This larger range in observed increases during wet weather (relative to the main stem areas) indicates a larger influence of point source discharges within the tributaries. No wet-weather data were reported for Whale Creek. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **English Kills**. TPAH (34) concentrations range from 0.11 to 7.3 µg/L (arithmetic average = 1.7 µg/L; median = 1.2 µg/L).
  - **East Branch**. TPAH (34) concentrations range from 0.093 to 4.3 µg/L (arithmetic average = 1.8 µg/L; median = 1.5 µg/L).
  - **Maspeth Creek**. TPAH (34) concentrations range from 0.24 to 6.2 µg/L (arithmetic average = 2.6 µg/L; median = 2.3 µg/L).
  - **Dutch Kills**. TPAH (34) concentrations range from 0.21 to 9.3 µg/L (arithmetic average = 2.2 µg/L; median = 0.97 µg/L).

4.7.4.2.2 Comparison Between Round 1 and Round 2 Sampling

Figure 4-181 shows that TPAH (34) concentrations in samples collected during Round 1 are generally higher than those collected during Round 2 for most individual wet weather sampling events, consistent with results from TPAH (17). In four of the five events, the number of values below the 1:1 line was greater than the number of values above (indicating that Round 1 values are generally higher than Round 2). The event in which there were more values above the 1:1 line had a p-value greater than 0.05; therefore, it is not significant. Deep sample results (denoted as squares in Figure 4-181) were often clustered more closely around the 1:1 line, indicating that the deep sample concentrations from Rounds 1 and 2 were similar. This pattern indicates a decrease in near-surface concentrations between the time of discharge and the next day, which is suggestive of mixing and deposition processes. This pattern was not observed for all five sampling events, indicating variability associated with amount, intensity, and timing of precipitation.

4.7.4.3 TPCB

4.7.4.3.1 Spatial Distribution

TPCB concentrations in surface water measured during wet weather sampling events from each reach of the Study Area are summarized in Table 4-58; a box plot showing the wet weather data (compared to dry weather) by reach is provided in Figure 4-182.

The range, arithmetic average, and median (TPCB congener concentrations) for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1, CM 1 – 2, and CM 2+**. Similar to the patterns observed for TPAH (17) and TPAH (34), wet weather TPCB concentrations show evidence of an increase with distance upstream in these main stem reaches, with the lowest arithmetic average concentrations in the area closest to the East River (CM 0 – 1), suggesting an increased effect from point sources with distance upstream. However, a single high concentration of 120 ng/L was also observed in CM 0 – 1. Arithmetic average wet weather TPCB concentrations in these main stem areas range from 1.5 to 2 times higher than the corresponding dry weather concentrations. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **CM 0 – 1**. TPCB concentrations range from 3.0 to 120 ng/L (arithmetic average = 11 ng/L; median = 7.8 ng/L).
  - **CM 1 – 2**. TPCB concentrations range from 4.5 to 46 ng/L (arithmetic average = 14 ng/L; median = 13 ng/L).
  - **CM 2+**. TPCB concentrations range from 5.0 to 61 ng/L (arithmetic average = 16 ng/L; median = 13 ng/L).
- **English Kills**. TPCB concentrations range from 2.7 to 57 ng/L (arithmetic average = 23 ng/L; median = 21 ng/L). The highest wet weather TPCB concentrations in the Study Area are observed within this tributary, where the average TPCB concentration is 1.4 to 2.1 times higher than the arithmetic average concentration in the main stem and 1.6 to 2.7 times higher than any of the other tributaries. Further, the arithmetic average wet weather concentration in English Kills is approximately 1.5 times higher than the dry weather average in this area.

- **East Branch, Maspeth Creek, and Dutch Kills**. Average wet weather concentrations in these tributaries are somewhat lower than those observed in English Kills; however, they are generally consistent with average wet weather concentrations in nearby main stem areas. Wet weather concentrations in these areas range from 1.5 to 2 times higher than dry weather concentrations. Ranges, arithmetic averages, and medians of concentrations are as follows:
    - **East Branch**. TPCB concentrations range from 2.4 to 30 ng/L (arithmetic average and median = 14 ng/L).
    - **Maspeth Creek**. TPCB concentrations range from 4.5 to 20 ng/L (arithmetic average and median = 11 ng/L).
    - **Dutch Kills**. TPCB concentrations range from 3.0 to 34 ng/L (arithmetic average = 8.5 ng/L; median = 6.7 ng/L).

4.7.4.3.2 Comparison Between Round 1 and Round 2 Sampling

Figure 4-182 shows that TPCB concentrations in samples collected during Round 1 are generally higher than those collected during Round 2. Figure 4-183 further evaluates differences between Round 1 and Round 2 sampling at all locations, by comparing TPCB concentrations measured in Round 1 versus Round 2 (cross plots) for each individual wet weather sampling event. This figure also shows that Round 1 concentrations are generally higher than those collected in Round 2 for most events (in all five events, the number of values below the 1:1 line were greater than the number of values above, although binomial test p values were < 0.05 for one event, with the others ranging from 0.07 to 0.3). However, there is variability among events, with concentrations between the two rounds of sampling tending to be most similar in Events 1 and 3. Additional discussion of the Round 1 and Round 2 surface water data, including comparisons to rainfall amounts, is provided in Section 6.4.2.2.

4.7.4.4 2,3,7,8-TCDD

2,3,7,8-TCDD concentrations in surface water measured during wet weather sampling events from each reach of the Study Area are summarized in Table 4-59; a box plot showing the wet weather data (compared to dry weather) by reach is provided in Figure 4-184.

Similar to the 2,3,7,8-TCDD concentrations in surface water measured during dry weather sampling events, the vast majority of 2,3,7,8-TCDD measured during wet weather sampling events had results less than the MDL. Only 3 of 192 wet weather surface water samples had detected 2,3,7,8-TCDD (one sample from CM 0 – 1 and two samples from CM 1 – 2). Therefore, similar to the dry weather sampling events, there are no spatial and temporal patterns of 2,3,7,8-TCDD in surface water under wet weather conditions within the limits of detection. The variability in the boxes in Figure 4-184 is due to variation in the MDL for this analyte.

4.7.4.5 Cu

4.7.4.5.1 Spatial Distribution

Concentrations of Cu in surface water measured during wet weather sampling events from each reach of the Study Area are summarized in Table 4-60; a box plot showing the wet weather data (compared to dry weather) by reach is provided in Figure 4-185.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1, CM 1 – 2, and CM 2+**. Similar to the patterns observed in TPAH (17), TPAH (34), and TPCB, wet weather Cu concentrations show evidence of an increase with distance upstream in these main stem reaches, with the lowest concentrations in the area closest to the East River (CM 0 – 1). This suggests an increased effect from point sources with distance upstream. Wet weather Cu concentrations in CM 0 – 1 are only slightly higher than the dry weather concentrations in this area. However, arithmetic average wet weather Cu concentrations in CM 1 – 2 and CM 2+ are approximately 1.5 times higher than the corresponding dry weather concentrations, with the maximum concentration within the Study Area observed in CM 2+. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **CM 0 – 1**. Cu concentrations range from 2.0 to 12 µg/L (arithmetic average = 4.9 µg/L; median = 4.6 µg/L).
  - **CM 1 – 2**. Cu concentrations range from 2.0 to 15 µg/L (arithmetic average = 6.2 µg/L; median = 6.3 µg/L).
  - **CM 2+**. Cu concentrations range from 2.0 to 110 µg/L (arithmetic average = 10 µg/L; median = 7.0 µg/L).

- **Tributaries**. Arithmetic average wet weather Cu concentrations in the tributaries are generally 1.5 to 2 times higher than those observed in CM 0 – 1 (with the exception of Dutch Kills); however, wet weather concentrations in the tributaries are generally comparable to those observed in CM 1 – 2 and CM 2+. Arithmetic average wet weather Cu concentrations in the tributaries range from 1.5 to 2 times higher than the corresponding dry weather concentrations. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **English Kills**. Cu concentrations range from 2.0 to 21 µg/L (arithmetic average = 7.6 µg/L; median = 7.7 µg/L).
  - **East Branch**. Cu concentrations range from 2.0 to 20 µg/L (arithmetic average = 7.4 µg/L; median = 7.3 µg/L).
  - **Maspeth Creek**. Cu concentrations range from 2.0 to 14 µg/L (arithmetic average = 8.2 µg/L; median = 8.8 µg/L).
  - **Dutch Kills**. Cu concentrations range from 2.0 to 26 µg/L (arithmetic average = 8.0 µg/L; median = 4.8 µg/L).

4.7.4.5.2 Comparison Between Round 1 and Round 2 Sampling

Figure 4-185 shows that contrary to TPAH (17), TPAH (34), and TPCB, median Cu concentrations measured during Round 1 tend to be somewhat lower than those measured during Round 2. Figure 4-186 further evaluates differences between Round 1 and Round 2 sampling at all locations by comparing detected Cu concentrations measured in Round 1 versus Round 2 (cross plots) for each individual wet weather sampling event. This figure shows that when Cu is detected in both rounds, the concentrations are highly variable. With the exception of Event 5, detected Cu concentrations in Round 1 and Round 2 (shallow or deep) were scattered around the 1:1 line, indicating no significant temporal pattern or pattern with depth in the Cu concentrations. Event 5 is the only sampling event where a majority of sample results were greater in Round 2 than in Round 1 (all but two samples; p value from binomial test < 0.05).

Figure 4-186 only includes detected Cu results, whereas Figure 4-185 includes detected Cu results and non-detect results (set to the MDL). Inclusion of the large number of non-detect results leads to the general observation that Round 1 Cu concentrations are variable and overlap to a great extent, but are somewhat less than those in Round 2 for certain reaches and tributaries (see Figure 4-185), which is the opposite of the observation for TPAH (17), TPAH (34), and TPCB. Although they may share the same underlying physical processes, wet weather results can vary by chemical, given differences in sources, including temporal variation in discharge concentration during events (which is related to spatial variations in the sewershed), as well as the relative strength of differing sources (e.g., point source discharge versus localized sediment erosion; see Section 6.3.3).

4.7.4.6 Pb

4.7.4.6.1 Spatial Distribution

Concentrations of Pb in surface water measured during wet weather sampling events from each reach of the Study Area are summarized in Table 4-61; a box plot showing the wet weather data (compared to dry weather) by reach is provided in Figure 4-187.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1, CM 1 – 2, and CM 2+**. Wet weather Pb concentrations show some evidence of an increase with distance upstream in these main stem reaches, with the lowest concentrations in the area closest to the East River (CM 0 – 1). Wet weather concentrations in CM 1 – 2 and CM 2+ are similar. Comparisons to dry weather Pb concentrations are confounded by the low FoD and higher detection limits in some dry weather samples. Ranges, arithmetic averages, and medians of wet weather concentrations are as follows:
  - **CM 0 – 1**. Pb concentrations range from 2.0 to 4.4 µg/L (arithmetic average = 2.5 µg/L; median = 2.0 µg/L).
  - **CM 1 – 2**. Pb concentrations range from 2.0 to 9.2 µg/L (arithmetic average = 3.4 µg/L; median = 2.0 µg/L).
  - **CM 2+**. Pb concentrations range from 2.0 to 380 µg/L (arithmetic average = 16 µg/L; median = 3.9 µg/L).
- **Tributaries**. Arithmetic average wet weather Pb concentrations in the tributaries are generally 1.4 to 2.8 times higher than those observed in CM 0 – 1. Wet weather

concentrations in East Branch and English Kills are generally comparable to those observed in CM 1 – 2 and CM 2+, while concentrations in Maspeth Creek are slightly greater than the upstream main stem reaches. Comparisons to dry weather Pb concentrations are confounded by the aforementioned dry weather detection issues. Ranges, arithmetic averages, and medians of concentrations are as follows:

- **English Kills**. Pb concentrations range from 2.0 to 86 µg/L (arithmetic average = 7.0 µg/L; median = 4.0 µg/L).
- **East Branch**. Pb concentrations range from 2.0 to 16 µg/L (arithmetic average = 4.6 µg/L; median = 3.8 µg/L).
- **Maspeth Creek**. Pb concentrations range from 2.0 to 11 µg/L (arithmetic average = 4.6 µg/L; median = 4.4 µg/L).
- **Dutch Kills**. Pb concentrations range from 2.0 to 12 µg/L (arithmetic average = 3.6 µg/L; median = 2.0 µg/L).

4.7.4.6.2 Comparison Between Round 1 and Round 2 Sampling

Figure 4-188 shows differences between Round 1 and Round 2 sampling at all locations by comparing detected Pb concentrations measured in Round 1 versus Round 2 (cross plots) for each individual wet weather sampling event. This figure shows that when Pb is detected in both rounds, the concentrations are variable, although the highest concentrations in Events 1 and 3 were observed in the Round 1 samples. With the exception of Events 4 and 5, for which fewer than two paired samples had detected Pb concentrations, Pb concentrations in Round 1 and Round 2 (shallow or deep) were generally scattered around the 1:1 line, indicating no significant temporal pattern or pattern with depth in the Pb concentrations.

4.7.5 Particulate Phase Concentrations

Particulate phase concentrations for chemicals with a strong affinity for solids can represent a significant fraction of the total water column concentration and are also useful for comparing to surface sediment concentrations. Based on the calculations summarized in Section 4.1.3.5 and outlined in detail in Attachment E-C of Appendix E, estimated and calculated particulate phase results for the surface water sampling programs are described in the following subsections. These results help to provide insight into how sources of solids and chemicals to the system are influencing surface sediment concentrations in the Study Area, as discussed in Sections 6 and 8.

4.7.5.1 TPAH (17), TPAH (34), and TPCB

Summary statistics for estimated particulate phase TPAH (17), TPAH (34), and TPCB concentrations in the Study Area and Phase 2 reference areas for samples collected during dry weather are presented in Tables 4-62, 4-63, and 4-64, respectively. Similar summary statistics for estimated particulate phase TPAH (17), TPAH (34), and TPCB concentrations in wet weather samples are provided in Tables 4-65, 4-66, and 4-67, respectively. The dry and wet weather particulate phase concentration data are also shown spatially in box plots in Figures 4-189 through 4-191.

In summary, spatial patterns in particulate phase TPAH (17), TPAH (34), and TPCB concentrations show similar patterns to those observed in whole-water samples described in the previous subsections; specifically, these patterns are as follows:

- The lowest concentrations tend to be in CM 0 – 1 (i.e., the CM nearest the East River).
- Concentrations tend to increase with increasing distance upstream in the main stem of Newtown Creek.
- Concentrations generally tend to be higher in the more upstream tributaries.

These patterns tend to be more prevalent during wet weather conditions.

Also, there are differences between Study Area particulate phase concentrations and those in the reference areas. For example, when the arithmetic average particulate phase TPAH (17), TPAH (34), and TPCB concentrations are averaged across the eight Study Area reaches, the concentrations are 3.2, 2.7, and 7.2 times higher than values for the arithmetic average concentrations averaged across the four Phase 2 reference areas under dry weather conditions, respectively. Under wet weather conditions, the arithmetic average particulate phase TPAH (17), TPAH (34), and TPCB concentrations, when averaged across the eight Study Area reaches, are 7.6, 9.5, and 16 times greater, respectively, than the arithmetic average for dry weather conditions, when averaged across the four Phase 2 reference areas (recognizing that no wet weather sampling data were collected from the reference areas).

Particulate phase concentrations also show similar increases in wet weather concentrations relative to dry weather conditions. For example, on a Study Area-wide basis, arithmetic average dry weather particulate phase TPAH (17), TPAH (34), and TPCB concentrations are 3.9, 9.5, and 0.32 mg/kg, respectively. These arithmetic average concentrations increase to 14, 31, and 0.72 mg/kg, respectively, under wet weather conditions. These increases represent approximate 2- to 4-fold increases in wet weather concentrations over dry weather concentrations (similar to the range of increases observed in whole-water samples).

4.7.5.2 2,3,7,8-TCDD

Particulate phase calculations for 2,3,7,8-TCDD are not presented in this report, because site-specific partitioning calculations were not possible, given that 2,3,7,8-TCDD was not analyzed in porewater samples. Regardless, due to the paucity of detectable 2,3,7,8-TCDD concentrations in both dry and wet weather surface water samples (see Sections 4.7.3.4.1 and 4.7.4.4.1), such calculations would be highly uncertain and would not likely provide any useful information.

4.7.5.3 Cu and Pb

Summary statistics for calculated particulate phase Cu and Pb concentrations for dry weather samples are presented in Tables 4-68 and 4-69, respectively. Summary statistics for calculated particulate phase Cu and Pb concentrations for wet weather samples are presented in Tables 4-70 and 4-71, respectively. Dry and wet weather box plots of calculated particulate phase Cu and Pb concentrations are shown in Figures 4-192 and 4-193, respectively.

Spatial patterns in particulate phase Cu and Pb concentrations show similar patterns to those observed in whole-water samples, as well as the particulate phase TPAH (17), TPAH (34), and TPCB concentrations described in Section 4.7.5.1; specifically, these patterns are as follows:

- The lowest concentrations tend to be in CM 0 – 1 (i.e., the CM nearest the East River).
- Concentrations tend to increase with increasing distance upstream in the main stem of Newtown Creek and in the tributaries.

These patterns tend to be more prevalent during wet weather conditions. Also, arithmetic average particulate phase Cu and Pb concentrations across the reaches of the Study Area range from being slightly lower to being up to 3.9 times greater than the arithmetic averages observed in the reference areas under dry weather conditions. During wet weather conditions, the average particulate phase Cu and Pb concentrations across the Study Area reaches range from being approximately 3 times lower to 33 times greater than the arithmetic average concentrations during dry weather conditions across the four Phase 2 reference areas (recognizing that no wet weather sampling data were collected from the reference areas).

Particulate phase concentrations for Cu and Pb also show similar increases in wet weather concentrations relative to dry weather conditions. For example, arithmetic average wet weather particulate phase Cu concentrations are 1.3 to 3.6 times greater than the dry weather concentration by reach. Similarly, arithmetic average wet weather particulate phase Pb concentrations are 1.2 to 16 times greater than the dry weather concentrations by reach, with the exception of CM 0 – 1 (the wet weather arithmetic average is slightly less than during dry weather) and Dutch Kills (equal averages in dry and wet weather). These concentration increases are similar to the observed increases in whole-water samples.

4.8 Porewater

4.8.1 Porewater Dataset

Porewater is water that occupies the interstitial pore space of aquatic sediment. Porewater originates as surface water from above or groundwater from below, and may represent a mixture of those two waters; the relative amounts of each depend on rates of groundwater movement and tidal exchange. Porewater sampling in the Study Area and Phase 2 reference areas was conducted to support the following:

- The nature and extent evaluation (this section), by providing data on aqueous phase chemical concentrations in the surface and subsurface sediment
- The BERA (see Section 4.2.4.3.2 of Appendix I), by directly measuring the chemical concentrations in surface sediment porewater to which benthic organisms are exposed, as opposed to back-calculating such potential exposure concentrations from bulk sediment concentrations
- The CFT evaluations (see Section 6.4), by providing a means to evaluate site-specific partitioning between chemicals in sediment in order to quantify the potential exchange of dissolved chemicals between sediment and water

- The development of the CSM (see Section 8), by incorporating data from a medium that is influenced by a variety of sources of COPCs to the Study Area and that is important for assessing potential risk, as well as fate and transport

Data were collected throughout the Study Area and Phase 2 reference areas as part of two sampling programs: the BERA triad program and the RI groundwater investigation program. Porewater samples were collected from sediment depth intervals of 0 to 15 cm (0 to 6 inches) and 15 to 30 cm (6 to 12 inches). These two depth intervals are together referred to in this section as *shallow* porewater. Porewater samples were also collected from *mid-depth* within the subsurface sediment, at the approximate midpoint between the mudline and underlying native material.

The shallow porewater samples were collected using passive sampling methods (SPME for organics and peepers for metals, both in situ and ex situ; see Section 4.2.2 of the Phase 2 FSAP Volume 1 [Anchor QEA 2014c] for more detail on these methods). The mid-depth porewater samples were collected using low-flow sampling techniques from temporary wellpoints installed within the sediment, at depths ranging from 2 to 9 feet below the mudline. The BERA triad program included 0- to 15-cm (0- to 6-inch) interval sediment samples that were sampled for porewater with passive samplers deployed in the laboratory, in conjunction with the 28-day sediment toxicity tests. The groundwater program included samples from all three depths (i.e., 0 to 15 cm [0 to 6 inches], 15 to 30 cm [6 to 12 inches], and mid-depth), and were sampled in the field. The in situ passive sampling results represent a temporal average over the duration of sampler deployment (which was 6 weeks; see Section 6.1.1 of Appendix Bi) for the 0- to 15-cm (0- to 6-inch) and 15- to 30-cm (6- to 12-inch) intervals, while the mid-depth samples represent a single point in time, at the time of collection.

There are advantages, disadvantages, and potential limitations associated with any porewater sampling technique (e.g., USEPA 2001). With respect to the SPME passive sampling method, shallow porewater chemical concentrations measured using this method are affected by sediment sorption/desorption (i.e., partitioning) processes, and, when used in situ, they can be affected by dissolved phase processes, including porewater advection and tidal exchange that result from reversals in seepage direction. The potential limitations or uncertainties these differing processes may introduce to measurements of shallow porewater concentrations are dependent on the factors affecting partitioning behavior, as well as the significance of dissolved phase processes relative to sediment sorption/desorption for in situ measurements (i.e., if tidal exchange occurs much more rapidly than sorption/desorption, short-term variations in chemical concentration may occur—although their net effect would be integrated into the long-term average measured by an in situ passive sampler). More discussion on the potential uncertainties of the shallow porewater chemical data are provided elsewhere in this report as follows:

- With respect to partitioning behavior, porewater concentration may not always be directly proportional to bulk sediment concentration (and in fact there can be cases where porewater concentration relative to collocated sediment concentration is much higher or much lower than expected from standard literature partitioning relationships). This variability in partitioning behavior can be due to effects from several factors, including the following:
    - Multiple sources and forms of OC in the sediment, including those associated with anthropogenic hydrocarbons (see Section 4.2.2.4)
    - A fraction of the sediment OC being the same hydrocarbons that are being evaluated (e.g., PAHs)
    - The presence of NAPL in sediment, which interacts with porewater differently than contaminants sorbed to the solid matrix

  These factors are discussed in detail as part of an evaluation of site-specific partitioning relationships in Section 6.4.1.3 (see specifically Sections 6.4.1.3.3 and 6.4.1.3.4).[126] Partitioning relationship evaluations are also being performed as part of the CFT model being developed to support the FS and will continue to be evaluated going forward.

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[126] The site-specific evaluations show that partitioning of contaminants to sediment in the Study Area is strong, but the process is complex and not entirely controlled by the total OC (unlike many other sites) due to the influence of multiple sources and forms of OC in Study Area sediment. The evaluations in Section 6.4.1.3.4 indicate it is likely that Study Area OC is a complex mixture of natural organic matter and anthropogenic OC sources, including soot carbon (and related materials) and weathered residual NAPL. Organic contaminants sorb to these different forms of OC to varying degrees; as such, they likely all serve as sorbents to some extent in Newtown Creek but with none dominating partitioning.

- Discussion of the relative importance of dissolved versus sediment phase processes and their potential influence on in situ porewater measurements, including data-based evaluations, is presented in Section 6.4.3.1.[127]
- Furthermore, SPME-based sampling methods measure only the freely dissolved phase (USEPA 2012d) and do not account for chemicals present in the DOC-bound phase, which can be a component of the total dissolved chemical concentration for organic contaminants in porewater. For PAHs and PCBs, the concentration of the DOC-bound phase is relatively small compared to the freely dissolved phase in porewater based on site-specific porewater DOC concentrations (see Sections 6.4.1.3.3 and 6.4.3.1.1 for additional discussion of this phase).

With respect to the temporary wellpoints used for mid-depth porewater, such extraction-based techniques may have difficulty achieving true phase separation, due to entrainment of solids in the samples (this limitation is discussed further in Sections 4.8.3 and 4.9).

Additional details on the sampling programs and sampling methodologies are provided in Section 6 of Appendix Bi. The data from the shallow and mid-depth porewater sampling are presented in Sections 4.8.2 and 4.8.3, respectively.

4.8.2 Shallow Porewater

4.8.2.1 Salinity

Porewater salinity is useful to understand the nature and source of sampled water, because porewater within the surface sediment of a coastal aquatic system can represent a mixture of more saline water from tidal surface water, and freshwater from groundwater in locations where it is discharging to the surface water.[128] Salinity was not analyzed in the shallow porewater samples during the Phase 2 groundwater sampling program, so only data from the

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[127] Based on the results of these evaluations (which included multiple lines of empirical evidence), it was concluded that shallow porewater chemical concentrations are largely controlled by partitioning with sediments (i.e., sorption/desorption effects) and that tidal exchange and porewater advection have, at most, secondary impacts on the observed concentrations. The CFT model being developed to support the FS will further evaluate these processes as well.

[128] Given historical saltwater intrusion into the surficial aquifer system within the Study Area, it is recognized that discharging groundwater may contain elevated salinity (see Section 4.9.2), so salinity alone may not be able to distinguish the origin of shallow porewater.

triad program, in which samples were collected and transported for processing in the laboratory, were available to characterize shallow porewater salinity. As such, these represent a snapshot of the porewater conditions at the time the triad sediment samples were collected. However, given that salinity behaves conservatively, the laboratory-based measurements are considered equivalent to other in situ field measurements of salinity (e.g., those for surface water).

Salinity in shallow porewater samples from the Study Area ranges from 3.7 to 22 psu, with an arithmetic average of 18 psu (see Table 4-72). For reference, the salinity of surface water in the Study Area is typically in the range of 20 to 25 psu, during dry weather conditions. Arithmetic average salinity values for shallow porewater in CM 0 – 1 and CM 1 – 2 (21 and 20 psu, respectively) are similar to surface water, suggesting East River influence. The arithmetic average salinity for shallow porewater in other reaches is lower, indicating varying degrees of freshwater inflow. The arithmetic average salinity in CM 2+ is 16 psu, while the arithmetic averages in Maspeth Creek, East Branch, and English Kills are in the range of 10 to 20 psu. The arithmetic average salinity in Dutch Kills is 15 to 20 psu at locations near its mouth and midway up the tributary, with a locally low value of 3.7 psu observed at the one location at its head (see Figures 4-194 and 4-195). Salinity of porewater collected from the four Phase 2 reference areas varies by reference area, and is within the range or slightly higher than that of the Study Area.

To assess the potential influence of tidal exchange (see Section 4.8.1) on shallow porewater salinity, the tide stage at the time of triad sample collection is indicated in Figure 4-195, based on the approach for defining low tide, high tide, and median water described in Section 4.7.2.1. This figure shows that the triad shallow porewater samples were collected under a range of tide stages, but no consistent relationship between shallow porewater salinity and tide stage was observed within the Study Area. Based on this observation and given the relatively uniform longitudinal gradient (see Figure 4-195), the shallow porewater salinity pattern from the triad dataset is considered representative of average conditions.

4.8.2.2 Distribution of Contaminants

Spatial distributions of TPAH (17), TPAH (34), TPCB, Cu, and Pb in shallow porewater are presented in the following subsections. C19-C36 and 2,3,7,8-TCDD were not evaluated here, because shallow porewater samples were not analyzed for these chemicals.

4.8.2.2.1 TPAH (17) Spatial Distribution

TPAH (17) concentrations in shallow porewater samples collected from each reach of the Study Area and the four Phase 2 reference areas are summarized in Table 4-73; a plan view map and longitudinal profile showing the data are provided in Figures 4-196 and 4-197, respectively. Concentrations in CM 0 – 2 are generally within the range of the data from the four Phase 2 reference areas but are observed to be somewhat higher elsewhere in the Study Area.

The range, arithmetic average, and median for each reach and notable spatial patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 2**. Shallow porewater TPAH (17) concentrations in CM 0 – 1 and CM 1 – 2 are generally the lowest in the Study Area. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **CM 0 – 1**. TPAH (17) concentrations range from 0.035 to 2.7 µg/L (arithmetic average = 0.60 µg/L; median = 0.29 µg/L).
  - **CM 1 – 2**. TPAH (17) concentrations range from 0.06 to 0.50 µg/L (arithmetic average = 0.24 µg/L; median = 0.22 µg/L).
- **CM 2+**. TPAH (17) concentrations range from 0.34 to 22 µg/L (arithmetic average = 4.4 µg/L; median = 1.8 µg/L). The highest concentrations in the main stem are in CM 2+.
- **English Kills**. TPAH (17) concentrations range from 0.47 to 52 µg/L (arithmetic average = 12 µg/L; median = 3.6 µg/L). Concentrations in the lower portion of English Kills are the highest in the Study Area. From there, concentrations decline upstream to the head of the tributary.
- **East Branch and Maspeth Creek**. Concentrations in these two tributaries are within the lower end of the range of concentrations measured in CM 2+ and show relatively little spatial variation within the tributaries. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **East Branch**. TPAH (17) concentrations range from 0.59 to 2.9 µg/L (arithmetic average = 1.3 µg/L; median = 1.1 µg/L).
  - **Maspeth Creek**. TPAH (17) concentrations range from 0.79 to 2.5 µg/L (arithmetic average and median were not calculated because only one sample had a detectable result).

- **Dutch Kills and Whale Creek**. Concentrations near the head of Dutch Kills are two of the highest shallow porewater TPAH (17) measurements in the Study Area. Concentrations at the other locations within Dutch Kills, as well as Whale Creek, are more similar to the adjacent main stem. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **Dutch Kills**. TPAH (17) concentrations range from 0.81 to 26 µg/L (arithmetic average = 7.0 µg/L; median = 1.1 µg/L).
  - **Whale Creek**. TPAH (17) concentrations range from 1.2 to 2.2 µg/L (arithmetic average = 1.7 µg/L; median = 1.7 µg/L).

4.8.2.2.2 TPAH (34) Spatial Distribution

TPAH (34) concentrations in shallow porewater samples collected from each reach of the Study Area and the four Phase 2 reference areas are summarized in Table 4-74; a plan view map and longitudinal profile showing the data are provided in Figures 4-198 and 4-199, respectively. Concentrations in CM 0 – 2 are generally within the range of the data from the four Phase 2 reference areas but are observed to be higher elsewhere in the Study Area. The arithmetic averages of TPAH (34) shallow porewater concentrations from CM2+ and English Kills are much higher than the arithmetic averages from the other Study Area reaches.

The range, arithmetic average, and median for each reach are as follows (no notable patterns were identified within any reach, although differences between reaches are observed):

- **CM 0 – 2.** Shallow porewater TPAH (34) concentrations in CM 0 – 1 and CM 1 – 2 are generally the lowest in the Study Area. The range, arithmetic average, and median for each reach are as follows:
  - **CM 0 – 1**. TPAH (34) concentrations range from 0.035 to 4.5 µg/L (arithmetic average = 1.1 µg/L; median = 0.51 µg/L).
  - **CM 1 – 2**. TPAH (34) concentrations range from 0.06 to 2.5 µg/L (arithmetic average = 0.62 µg/L; median = 0.45 µg/L).
- **CM 2+**. TPAH (34) concentrations in CM2+ are greater than those in CM 0 – 2, especially upstream of CM 2.3, and range from 0.67 to 240 µg/L (arithmetic average = 45 µg/L; median = 8.9 µg/L).

- **English Kills**. Shallow porewater concentrations of TPAH (34) are the highest of any recorded in the Study Area. TPAH (34) concentrations range from 1.7 to 620 µg/L (arithmetic average = 110 µg/L; median = 28 µg/L). The highest concentrations in English Kills are present around CM 3.0 to 3.4; values are lower near the head of the tributary, but they generally exceed those in CM 0 – 2 by a factor of 10.
- **East Branch and Maspeth Creek.** Concentrations in these two tributaries are within the lower end of the range of concentrations measured in CM 2+. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **East Branch**: TPAH (34) concentrations range from 1.3 to 6.5 µg/L (arithmetic average = 3.7 µg/L; median = 3.3 µg/L).
  - **Maspeth Creek:** TPAH (34) concentrations range from 4.1 to 20 µg/L (arithmetic average = 10 µg/L; median = 6.1 µg/L).
- **Dutch Kills**. TPAH (34) concentrations range from 1.6 to 39 µg/L (arithmetic average = 13 µg/L; median = 3.7 µg/L) and are similar to or somewhat higher than those in the nearby main stem.
- **Whale Creek**. TPAH (34) concentrations range from 12 to 16 µg/L (arithmetic average = 14 µg/L; median = 14 µg/L) and are generally higher than those in the nearby main stem.

4.8.2.2.3 TPCB Spatial Distribution

TPCB concentrations in shallow porewater samples collected from each reach of the Study Area and the four Phase 2 reference areas are summarized in Table 4-75; a plan view map and longitudinal profile showing the data are provided in Figures 4-200 and 4-201, respectively. Shallow porewater TPCB concentrations throughout the Study Area exceed the data from the four Phase 2 reference areas.

The range, arithmetic average, and median for each reach and notable spatial patterns in the Study Area data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 2**. Shallow porewater TPCB concentrations in CM 0 – 1 and CM 1 – 2 exhibit relatively low variability and are generally the lowest in the Study Area. Ranges, arithmetic averages, and medians of concentrations are as follows:

  - **CM 0 – 1**. TPCB concentrations range from 4.2 to 9.0 ng/L (arithmetic average = 6.3 ng/L; median = 6.3 ng/L).
  - **CM 1 – 2**. TPCB concentrations range from 3.8 to 9.2 ng/L (arithmetic average = 6.1 ng/L; median = 6.0 ng/L).

- **CM 2+**. TPCB concentrations range from 7.0 to 110 ng/L (arithmetic average = 43 ng/L; median = 36 ng/L). Concentrations increase by approximately a factor of 10 going upstream in this reach. The highest concentrations in the main stem are in CM 2+.
- **English Kills**. TPCB concentrations range from 25 to 470 ng/L (arithmetic average = 140 ng/L; median = 79 ng/L). Concentrations in English Kills are the highest in the Study Area. The highest concentrations in English Kills are present around CM 3.0 to 3.3; values are lower near the head of the tributary, but still exceed those in CM 0 – 2 by a factor of 10.
- **East Branch**. TPCB concentrations range from 3.1 to 16 ng/L (arithmetic average = 7.5 ng/L; median = 6.5 ng/L). TPCB concentrations in this tributary are similar to those in CM 0 – 2 and are higher near its confluence with the main stem than at the locations closer to its head.
- **Maspeth Creek**. TPCB concentrations range from 2.6 to 7.4 ng/L (arithmetic average = 4.3 ng/L; median = 2.8 ng/L). Concentrations in this tributary are relatively low and are similar to, or lower than, those in CM 0 – 2.
- **Dutch Kills**. TPCB concentrations range from 2.6 to 11 ng/L (arithmetic average = 6.5 ng/L; median = 7.8 ng/L). Concentrations in Dutch Kills are similar to those in the adjacent main stem and relatively lower at its head, as compared to the other locations.
- **Whale Creek**. TPCB concentrations range from 13 to 15 ng/L (arithmetic average = 14 ng/L; median = 14 ng/L). Shallow porewater TPCB concentrations in this tributary are relatively low, but slightly higher than those in the adjacent main stem (i.e., arithmetic average is approximately double that of CM 0 – 1 and CM 1 – 2).

4.8.2.2.4 Cu Spatial Distribution

Cu concentrations in shallow porewater samples collected from each reach of the Study Area and the four Phase 2 reference areas are summarized in Table 4-76; a plan view map and longitudinal profile showing the data are provided in Figures 4-202 and 4-203, respectively. Shallow porewater Cu concentrations in all reaches overlap with the data from the four

Phase 2 reference areas, although there are also samples from most reaches (more often in the upper portion of the Study Area) that exceed the Phase 2 reference area range. Arithmetic average concentrations within the Study Area reaches exceed those in the Phase 2 reference areas by factors of approximately 3 to 7 (see Table 4-76). Cu concentrations are non-detect in several samples, and are relatively variable within the Study Area, potentially due in part to the differing sampling methods employed.[129] Due to this variability, there are no notable spatial patterns in the Study Area data, other than the tendency for samples in the upper portion of the Study Area (e.g., CM 2+) to be slightly higher than elsewhere.

The range, arithmetic average, and median for each reach are as follows (no notable patterns were identified in any reach):

- **CM 0 – 1**. Cu concentrations range from 0.32 to 9.0 μg/L (arithmetic average = 3.8 μg/L; median = 4.4 μg/L).
- **CM 1 – 2**. Cu concentrations range from 0.32 to 16 μg/L (arithmetic average = 4.7 μg/L; median = 3.0 μg/L).
- **CM 2+**. Cu concentrations range from 0.32 to 17 μg/L (arithmetic average = 9.0 μg/L; median = 8.6 μg/L).
- **English Kills**. Cu concentrations range from 0.32 to 18 μg/L (arithmetic average = 5.8 μg/L; median = 4.0 μg/L).
- **East Branch**. Cu concentrations range from 0.32 to 8.2 μg/L (arithmetic average = 4.1 μg/L; median = 4.0 μg/L).
- **Maspeth Creek**. Cu concentrations range from 0.32 to 3.3 μg/L (arithmetic average = 1.4 μg/L; median = 0.42 μg/L).
- **Dutch Kills**. Cu concentrations range from 0.32 to 8.4 μg/L (arithmetic average = 4.2 μg/L; median = 5.0 μg/L).

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[129] Unlike TPAH (17), TPAH (34), and TPCB, the shallow porewater data for Cu show some differences between the triad and groundwater programs. The data are similar in the upper portion of the Study Area but differ in CM 0 – 2, with the triad program data generally being lower than the groundwater program data by approximately a factor of 2 to 3 (see Figure 4-202). All data passed quality control requirements; the apparent difference between the triad and groundwater program Cu data may be due to differences in laboratories and sampling methods.

- **Whale Creek**. Cu concentrations range from 1.7 to 5.4 µg/L (arithmetic average = 3.5 µg/L; median = 3.5 µg/L).

4.8.2.2.5 Pb Spatial Distribution

Pb concentrations in shallow porewater samples collected from each reach of the Study Area and the four Phase 2 reference areas are summarized in Table 4-77; a plan view map and longitudinal profile showing the data are provided in Figures 4-204 and 4-205, respectively. Shallow porewater Pb concentrations in all reaches overlap with the data from the four Phase 2 reference areas. Arithmetic average concentrations within the Study Area reaches are generally similar to or slightly higher than those in the Phase 2 reference areas (see Table 4-77). Pb concentrations within the Study Area are variable, with approximately half of the samples being non-detect, potentially due in part to the differing sampling methods employed.[130] Due to this variability, there are no notable spatial patterns in the Study Area data, other than the tendency for the maximum sample values in the upper portion of the Study Area (e.g., CM 2+) to be slightly higher than elsewhere.

The range, arithmetic average, and median for each reach are as follows (no notable patterns were identified in any reach):

- **CM 0 – 1**. Pb concentrations range from 0.12 to 5.9 µg/L (arithmetic average = 2.8 µg/L; median = 4.0 µg/L).
- **CM 1 – 2**. Pb concentrations range from 0.12 to 4.0 µg/L (arithmetic average = 2.6 µg/L; median = 2.7 µg/L).
- **CM 2+**. Pb concentrations range from 0.08 to 7.4 µg/L (arithmetic average = 3.3 µg/L; median = 4.0 µg/L).
- **English Kills**. Pb concentrations range from 0.38 to 9.0 µg/L (arithmetic average = 4.1 µg/L; median = 4.0 µg/L).
- **East Branch**. Pb concentrations range from 0.17 to 5.5 µg/L (arithmetic average = 3.6 µg/L; median = 4.0 µg/L).

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[130] Similar to Cu, the shallow porewater data for Pb show some differences between the triad and groundwater programs, with a lower FoD for the groundwater program. All data passed quality control requirements; the apparent difference between the triad and groundwater program Pb data is due to differences in detection limits (the groundwater program employed a higher detection limit than the triad program) and also may be due to differences in laboratories and sampling methods.

- **Maspeth Creek**. Pb concentrations range from 0.54 to 9.4 µg/L (arithmetic average = 3.6 µg/L; median = 0.74 µg/L).
- **Dutch Kills**. Pb concentrations range from 0.16 to 4.0 µg/L (arithmetic average = 2.7 µg/L; median = 4.0 µg/L).
- **Whale Creek**. Cu concentrations range from 1.4 to 5.3 µg/L (arithmetic average = 3.3 µg/L; median = 3.3 µg/L).

4.8.2.3 Near-Surface Vertical Patterns

Near-surface vertical concentration distributions in shallow porewater throughout the Study Area were evaluated at the 15 groundwater program locations where samples were collected at two separate depth intervals (0 to 15 cm [0 to 6 inches] and 15 to 30 cm [6 to 12 inches]). Cross plots comparing concentrations from these two depth intervals at each location are presented in Figures 4-206 through 4-209 for TPAH (17), TPAH (34), TPCB, and Cu, respectively.[131,132] As shown in the cross plots, data from the two depth intervals trend with one another, and concentrations from the two depths vary by less than a factor of 2 in most cases. Concentrations of TPAH (17), TPAH (34), and TPCB tend to be higher in the 15- to 30-cm [6- to 12-inch] interval for a majority of locations, with two locations (one in the main stem and one in English Kills) indicating concentration differences greater than a factor of 2 for each of these three chemicals. For Cu, concentrations in the samples from the 0- to 15-cm [0- to 6-inch] depth were greater than those from the 15- to 30-cm [6- to 12-inch] depth for more than half of the locations, although all sample concentrations were within a factor of approximately 2 of one another. Evaluations of the processes responsible for these observed patterns (i.e., partitioning with sediments and dissolved phase transport via groundwater discharge and tidal exchange) are discussed further in Section 6.4.3.1.2.

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[131] These cross plots show detect data only. Therefore, Pb is not presented in this evaluation, as there were no pairs of samples from the 15 groundwater program locations in which Pb was detected in both samples.

[132] Comparisons of these data to data from mid-depth porewater and groundwater samples from the same locations are provided in Section 4.9.3.

4.8.3 Mid-Depth Porewater

Mid-depth porewater samples were collected from temporary groundwater monitoring wells installed at 17 locations throughout the Study Area,[133] with midpoints of the screened intervals at depths ranging from 2 to 9 feet below the sediment surface.[134] Initial evaluation of the analytical data showed elevated suspended solids and turbidity in most of these samples, which is a result of the fine-grained nature of the surrounding sediment. This was unavoidable, due to the well deployment methods used and the sampling technique employed in accordance with the USEPA *Final Groundwater Investigation Work Plan* (USEPA 2014a). Due to the high TSS in these samples, the laboratory-reported, whole-water (unfiltered) concentrations for organic constituents were converted to estimated dissolved phase concentrations (i.e., the phase that is available for transport in porewater) based on equilibrium partitioning theory. Details regarding the conversion, which was accepted by USEPA for use in the RI (Kwan 2016c), are provided in Section 3.7.2 of Appendix F. Due to the uncertainty associated with this conversion, the estimated dissolved phase concentrations reported here for organic constituents were determined using two methods. The first method (referred to hereafter as "Method 1") used dry weight partition coefficients derived from the analysis of Study Area shallow porewater data (described in Section 6.4.1.3, with values presented in Table 6-2b); the second method (referred to hereafter as "Method 2") used literature-based OC partition coefficients (described in Section 6.4.1.2, with values and reference citations presented in Table 6-1).[135] Both Method 1 and Method 2 were used for TPAH (17), TPAH (34), and TPCB. In general, estimated dissolved phase concentrations of

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[133] Mid-depth porewater samples were collected from each Study Area reach, except Whale Creek and Maspeth Creek.

[134] During sampling of the mid-depth sediment wells, 10 of the 17 wells went dry during low-flow sampling. The other mid-depth wells had 1 to 10 feet of drawdown. The mid-depth wells with small drawdown also had relatively low salinity; thus, the lack of drawdown does not indicate a preferential hydraulic connection to surface water at these locations. The low water yield from the sediment resulted in sample volume limitations that precluded analysis for some constituents. Specifically, whole-water chemical analyses were given priority over dissolved (field-filtered) metals and conventional parameters, including salinity and TSS. Of the 17 mid-depth porewater sample locations, the number of locations for which sufficient volume precluded analysis was 5 for salinity, 4 for TSS, 5 for dissolved Cu, and 5 for dissolved Pb (see Tables 4-78, 4-79, 4-85, and 4-86, respectively). Well development field records for mid-depth sediment porewater wells are included in Attachment Bi-C09 of Appendix Bi.

[135] Both methods used OC measurements (i.e., DOC for Method 1 and TOC and DOC for Method 2) from the mid-depth porewater samples; however, OC was non-detect at some locations, so OC from collocated sediment samples were used in Method 2 for some locations (see Section 3.7.2 of Appendix F for additional detail).

TPAH (17), TPAH (34), and TPCB derived from Method 2 are higher than those from Method 1. Evaluating concentrations derived from both methods provides a conservative range of estimates for the dissolved phase mid-depth porewater concentrations. C19-C36 was not analyzed in the shallow porewater data collection (see Section 4.8.2.2); therefore, estimates of dissolved phase concentrations via Method 1 could not be performed (i.e., there were no site-specific partitioning coefficients for C19-C36). Estimated dissolved phase concentrations for C19-C36 using Method 2 were developed and are presented in this section. For Cu and Pb, concentrations described in this section are based on dissolved results, which were measured analytically using field-filtered samples.

4.8.3.1 Salinity and TSS

Salinity in mid-depth porewater samples ranges from 11 to 22 psu, with an arithmetic average of 19 psu (see Table 4-78). The highest salinity values, which were in the range of 21 to 22 psu (i.e., within the typical range of the surface water), were reported between the mouth of Newtown Creek and CM 1.3, and at one location each from Dutch Kills and CM 2+ (see Figures 4-210 and 4-211). Salinity in the two remaining main stem samples upstream of CM 1.3 and in the three remaining tributary samples is generally lower, with a range of 11 to 21 psu and an arithmetic average of 15 psu. This spatial pattern is generally similar to that observed in the shallow porewater (i.e., values similar to surface water in the lower part of the main stem, with lower values in the upper tributaries; see Section 4.8.2.1).

TSS in the mid-depth porewater samples was reported at concentrations ranging from 27 to 2,100 mg/L, with an arithmetic average concentration of 240 mg/L (see Table 4-79). TSS concentrations vary significantly from sample to sample, with no apparent spatial trend (see Figures 4-212 and 4-213). TSS concentrations of this magnitude are elevated compared to what is typically observed for porewater, indicating that surrounding sediment was drawn into the samples during collection from the temporary wellpoints, as discussed previously in this section.

4.8.3.2 Distribution of Contaminants

4.8.3.2.1 TPAH (17)

Estimated dissolved phase TPAH (17) concentrations in mid-depth porewater samples are presented in Tables 4-80 and 4-81 and Figures 4-214a through 4-215b, which include results for both Method 1 and Method 2.[136] The data from the 13 samples[137] are variable and exhibit little spatial pattern other than a suggestion of decreasing concentration from upstream to downstream across the Study Area.

The ranges, arithmetic averages, and medians for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 2 and Dutch Kills**. Concentrations in these reaches are among the lowest in the Study Area and less than 10 µg/L in six of the eight samples (with the others being between 10 and 30 µg/L [Method 1] or 10 and 65 µg/L [Method 2]).
  - **CM 0 – 1**. TPAH (17) concentrations range from 0.18 to 11 µg/L for Method 1 and 0.54 to 14 µg/L for Method 2 (arithmetic averages of 4.6 and 6.4 µg/L, respectively; medians of 3.0 and 4.7 µg/L, respectively).
  - **CM 1 – 2**. TPAH (17) concentrations range from 0.43 to 29 µg/L for Method 1 and 3.3 to 63 µg/L for Method 2 (arithmetic averages of 10 and 24 µg/L, respectively; medians of 1.8 and 7.1 µg/L, respectively).
  - **Dutch Kills**. TPAH (17) concentrations range from 0.17 to 6.4 µg/L for Method 1 and 0.25 to 9.3 µg/L for Method 2 (arithmetic averages of 3.3 and 4.8 µg/L, respectively; medians of 3.3 and 4.8 µg/L, respectively).
- **CM 2+**. TPAH (17) concentrations range from 18 to 29 µg/L for Method 1 and 23 to 35 µg/L for Method 2 (arithmetic averages of 23 and 29 µg/L, respectively; medians of

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[136] Estimated dissolved phase TPAH (17), TPAH (34), and TPCB concentrations in mid-depth porewater samples were calculated based on whole-water concentration results, TSS, and site-specific (Method 1) and literature-based (Method 2) partition coefficients, as discussed in Section 3.7.2.1 of Appendix F. C19-C36 concentrations in mid-depth porewater samples were only calculated using literature-based (Method 2) partition coefficients, as there were no site-specific partition coefficients available for this chemical (see Section 4.8.3).

[137] Sample volume limitations prevented analysis for TSS in some samples (see Section 4.8.3 and Footnote 134). As a result, dissolved phase estimates were not developed for these locations (although the raw concentration data for these locations were generally similar to those for which dissolved phase estimates were developed; see Section 3.7.2 of Appendix F).

23 and 29 µg/L, respectively). The estimated dissolved phase TPAH (17) concentrations of the two samples in this reach are somewhat higher than in the lower reaches.

- **English Kills and East Branch**. The highest estimated dissolved phase TPAH (17) concentration in mid-depth porewater was reported in English Kills near CM 3.0. The concentrations of 2,000 µg/L for Method 1 and 4,800 µg/L for Method 2 in this sample are approximately 70 to 80 times larger than the next highest mid-depth porewater TPAH (17) concentration and is likely an overestimate due to potential interferences associated with NAPL in the sample.[138] The other two samples in these tributaries are low (less than 10 µg/L) and in the range of the CM 0 – 2 data.
  - **English Kills**. TPAH (17) concentrations range from 1.0 to 2,000 µg/L for Method 1 and 1.9 to 4,800 for Method 2 (arithmetic averages of 1,000 and 2,400 µg/L, respectively; medians of 1,000 and 2,400 µg/L, respectively).
  - **East Branch**. The TPAH (17) concentrations from the single mid-depth porewater sample collected in this reach are 4.3 and 6.8 µg/L for Method 1 and Method 2, respectively.

4.8.3.2.2 TPAH (34)

Estimated dissolved phase TPAH (34) concentrations in mid-depth porewater samples are presented in Tables 4-82a and 4-82b and Figures 4-216a through 4-217b, which include results for both Method 1 and Method 2. The data from the 13 samples are variable and exhibit little spatial pattern, other than a suggestion of decreasing concentration from upstream to downstream across the Study Area.

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[138] The estimated dissolved phase TPAH (17) concentration for this mid-depth porewater sample (at EK093) is considered an overestimate based on visual observations of sediment and effective solubility calculations. Visual observation of the sediment core collected at this location (core EK093SC-A) indicated potential NAPL at a depth of 90 to 91 cm (3 feet) (see Section 5.4.3.1 of Appendix C and Figures C5-20i and C5-20j of Appendix C), which is within the depth interval from which the mid-depth porewater sample was collected (15 to 107 cm) (0.5 to approximately 3.5 feet). Additionally, the calculations of effective solubility presented in Section 6 of Appendix F (and summarized in Section 4.9.3.1.1) were used to evaluate this mid-depth porewater sample and further confirmed the hypothesis that the estimated concentration was likely influenced by NAPL and is likely an overestimate of the actual dissolved phase value.

The range, arithmetic average, and median for each reach are as follows (no notable patterns were identified):

- **Main Stem**. Arithmetic mean TPAH (34) concentrations increase throughout the main stem as distance from the East River increases.
  - **CM 0 – 1.** TPAH (34) concentrations range from 0.47 to 13 µg/L for Method 1 and 1.3 to 16 µg/L for Method 2 (arithmetic averages of 6.4 and 8.7 µg/L, respectively; medians of 6.2 and 8.6 µg/L, respectively).
  - **CM 1 – 2**. TPAH (34) concentrations range from 1.3 to 45 µg/L for Method 1 and 15 to 86 µg/L for Method 2 (arithmetic averages of 20 and 39 µg/L, respectively; medians of 12 and 16 µg/L, respectively).
  - **CM 2+**. TPAH (34) concentrations range from 29 to 44 µg/L for Method 1 and 37 to 47 µg/L for Method 2 (arithmetic averages of 37 and 42 µg/L, respectively; medians of 37 and 42 µg/L, respectively).
- **Tributaries**.
  - **English Kills**. For both Method 1 and Method 2, TPAH (34) arithmetic mean concentrations in English Kills exceed values in all other reaches by two to three orders of magnitude. TPAH (34) concentrations range from 1.8 to 2,300 µg/L for Method 1 and 2.6 to 5,200 µg/L for Method 2 (arithmetic averages of 1,100 and 2,600 µg/L, respectively; medians of 1,100 and 2,600 µg/L, respectively). As discussed in Section 4.8.3.2.1 for TPAH (17), the mid-depth porewater concentration for TPAH (34) in the sample near CM 3.0 is also likely an overestimate due to potential interferences associated with NAPL in the sample.[138]
  - **East Branch**. The single TPAH (34) concentration in this reach is 13 µg/L for Method 1 and 15 µg/L for Method 2.
  - **Dutch Kills**. TPAH (34) arithmetic mean concentrations are lower than in any other reach (main stem or tributary). TPAH (34) concentrations range from 0.46 to 9.4 µg/L for Method 1 and 0.51 to 12 µg/L for Method 2 (arithmetic averages of 4.9 and 6.0 µg/L, respectively; medians of 4.9 and 6.0 µg/L, respectively).

4.8.3.2.3 C19-C36

Estimated dissolved phase C19-C36 concentrations in mid-depth porewater samples are presented in Table 4-83, Figure 4-218, and Figure 4-219 for Method 2 only.[139] Only 3 of the 13 samples had detected C19-C36 concentrations.[140] These three detections were at locations within the main stem, two in CM 1 – 2 (0.0076 and 0.022 mg/L, respectively) and one in CM 2+ (0.0081 mg/L). The MDLs for the ten non-detect samples varied, with the range of MDLs (0.00019 to 0.63 mg/L) being greater than the range of the three detected values. As such, given the uncertainty in the dissolved phase estimation method, these results likely indicate that C19-C36 is generally not present, or is present at low to undetectable concentrations, in mid-depth porewater.

4.8.3.2.4 TPCB

Estimated dissolved phase TPCB concentrations in mid-depth porewater samples are presented in Tables 4-84a and 4-84b and Figures 4-220a and 4-221b.[137] The data suggest a spatial pattern of decreasing concentration from upstream to downstream across the Study Area, although there is local variability.

The ranges, arithmetic averages, and medians for each reach and notable patterns in the data (based on both Method 1 and Method 2) are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **Main Stem and Dutch Kills**. Estimated dissolved phase TPCB concentrations in the majority of the samples from these reaches are less than or equal to 20 ng/L (10 of 10 samples for Method 1 and 8 of 10 samples for Method 2), with the other two samples

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[139] As discussed in the Section 4.8.3 introduction, there are no site-specific partitioning coefficients available for C19-C36, so dissolved concentrations via Method 1 could not be developed.

[140] As discussed in the Section 4.8.3 introduction, sample volume limitations precluded analysis for TSS in four of the mid-depth porewater sample locations. Calculation of dissolved phase contaminant concentrations requires TSS data (see Section 4.8.3), so estimated mid-depth dissolved phase contaminant porewater concentrations could not be calculated for 4 of the 17 mid-depth sample locations. For C19-C36, two of these four samples had detectable concentrations, meaning that there were five total samples with detected whole-water concentrations for C19-C36, although estimated dissolved phase porewater concentrations could only be developed for three of these samples. Evaluation of the whole-water concentrations of C19-C36 for the 17 mid-depth porewater locations does not change the data summary presented in this section.

being slightly higher (i.e., Method 2 concentrations at two locations from CM 1 – 2 in the range of 30 to 60 ng/L).

  - **CM 0 – 1**. TPCB concentrations range from 1.6 to 9.6 ng/L for Method 1 and 2.6 to 20 ng/L for Method 2 (arithmetic averages of 4.6 and 12 ng/L, respectively; medians of 2.6 and 14 ng/L, respectively).
  - **CM 1 – 2**. TPCB concentrations range from 2.3 to 30 ng/L for Method 1 and 5.8 to 60 ng/L for Method 2 (arithmetic averages of 12 and 34 ng/L, respectively; medians of 3.8 and 36 ng/L, respectively).
  - **CM 2+**. TPCB concentrations range from 2.3 to 3.6 ng/L for Method 1 and 1.8 to 2.8 ng/L for Method 2 (arithmetic averages of 3.0 and 2.3 ng/L, respectively; medians of 3.0 and 2.3 ng/L, respectively).
  - **Dutch Kills**. TPCB concentrations range from 4.2 to 16 ng/L for Method 1 and 3.3 to 8.2 ng/L for Method 2 (arithmetic averages of 9.9 and 5.7 ng/L, respectively; medians of 9.9 and 5.7 ng/L, respectively).

- **English Kills and East Branch**. The highest mid-depth porewater estimated dissolved phase TPCB concentration was reported in English Kills near CM 3.0, where concentrations (1,100 and 5,100 ng/L for Method 1 and Method 2, respectively) are estimated to be 10 to 54 times larger than the next highest concentrations, which are estimated at 110 ng/L (Method 1) and 94 ng/L (Method 2) observed in the sample from East Branch. The other sample from English Kills (concentrations of 18 and 94 ng/L for Method 1 and Method 2, respectively) was in the range of the data from the main stem.
  - **English Kills**. TPCB concentrations range from 18 to 1,100 ng/L for Method 1 and 13 to 5,100 ng/L for Method 2 (arithmetic averages of 580 and 2,500 ng/L, respectively; medians of 580 and 2,500 ng/L, respectively).
  - **East Branch**. The single sample from East Branch has estimated TPCB concentrations of 110 ng/L (Method 1) and 94 ng/L (Method 2).

4.8.3.2.5 Cu

Dissolved Cu concentrations in mid-depth porewater samples are presented in Table 4-85 and Figures 4-222 and 4-223. The data are variable, including being non-detect in several samples (5 of 12). They do not exhibit any spatial pattern.

The range, arithmetic average, and median for each reach are as follows (no notable patterns were identified):

- **Main Stem**. The two highest dissolved Cu concentrations, which are between 40 and 50 µg/L, are reported in the main stem at CM 0.08 and CM 1.4. The remaining concentrations (two each in CM 0 – 1 and CM 1 – 2, and one in CM 2+) are non-detect (three samples) or low (two samples less than 5 µg/L).
  - **CM 0 – 1**. Cu concentrations range from 2.0 to 45 µg/L (arithmetic average and median were not calculated because only one sample had a detectable result).
  - **CM 1 – 2**. Cu concentrations range from 2.0 to 44 µg/L (arithmetic average = 16 µg/L; median = 2.7 µg/L).
  - **CM 2+**. The single Cu concentration in this reach is 4.2 µg/L.
- **Tributaries**. The dissolved Cu concentrations of the tributary mid-depth porewater samples are non-detect (two samples in English Kills) or low (two samples in Dutch Kills and one in East Branch ranging from 4.5 to 6.0 µg/L).
  - **English Kills**. The two Cu concentrations in this reach are both 2.0 µg/L (both non-detect).
  - **East Branch**. The single Cu concentration in this reach is 4.5 µg/L.
  - **Dutch Kills**. Cu concentrations range from 5.2 to 6.0 µg/L (arithmetic average = 5.6 µg/L; median = 5.6 µg/L).

4.8.3.2.6 Pb

Dissolved Pb concentrations in mid-depth porewater samples are presented in Table 4-86 and Figures 4-224 and 4-225. The data are all non-detect, other than at two locations within the main stem (one location in CM 0 – 1 and another in CM 1 – 2). The dissolved Pb concentrations are these two locations are 28 µg/L and 34 µg/L, respectively. MDLs for the non-detect samples range from 2.0 to 4.0 µg/L.

4.9 Groundwater

4.9.1 Groundwater Dataset

Groundwater samples were collected throughout the Study Area from temporary monitoring wells installed in native material beneath the sediment, during the Phase 2 groundwater investigation. The temporary well depths ranged from 4.5 to 32 feet below the mudline, and 0.56 to 16 feet below the sediment/native material interface (top of screen depths). The temporary well screen lengths ranged from 2 to 5 feet. The temporary wells were installed following procedures presented in the Phase 2 FSAP Volume 2 Addendum No. 1 (Anchor QEA 2014g), which are also described in Section 6.3.1.3 of Appendix Bi. Each in-creek well was installed with a pre-pack well screen and bentonite seal attached to the riser above the screen to prevent short-circuiting of surface water into the wells. Field parameters and water levels were monitored during development and low-flow sampling and did not indicate any evidence of short-circuiting. Field sampling logs are provided in Attachment Bi-C09 of Appendix Bi.

All concentrations reported in this section are dissolved phase concentrations. Similar to the mid-depth porewater samples described in Section 4.8.3, dissolved Cu and Pb were measured in field-filtered samples. Also similar to the mid-depth porewater samples, laboratory-reported, whole-water (unfiltered) concentrations for organic constituents in groundwater samples were converted to dissolved phase concentrations (i.e., the phase that is available for transport in groundwater) using partitioning calculations. This conversion is necessary, because of elevated TSS in these samples. Details regarding the conversion to dissolved phase concentrations are provided in Section 3.7.2.1 of Appendix F.

4.9.2 Salinity, TDS, and TSS

4.9.2.1 Spatial Distribution

Salinity in groundwater samples from the native material ranges from non-detect (less than or equal to 2 psu) to 21 psu, with an arithmetic average of 9.1 psu (see Table 4-87). The highest salinity values were generally reported in samples collected between the mouth of Newtown Creek and CM 1.3 (see Figures 4-226 and 4-227) and are slightly less than the arithmetic average dry weather surface water salinity of 23 psu for the Study Area. The majority of samples collected upstream of CM 1.3 (34 of 41 samples) were below 10 psu (the 80th percentile), with more than half (24 of 41 samples; the 58th percentile) reported below 5 psu, but the upstream areas had a few exceptions where salinity values were higher. Section 5 presents information regarding historical groundwater pumping and saltwater intrusion in the region, which may have a residual effect on salinity in groundwater in certain areas. TDS concentrations in groundwater, which qualitatively match salinity trends in groundwater, range from 740 to 24,000 mg/L, and the arithmetic average TDS concentration is 9,700 mg/L (see Attachment F-H1 and Table F3-22 of Appendix F).

TSS in the groundwater samples were reported at an arithmetic average concentration of 320 mg/L and range from 14 to 3,600 mg/L (see Table 4-88). TSS concentrations vary significantly from sample to sample, with no apparent spatial trend (see Figures 4-228 and 4-229). As reported in Section 3.7.2 of Appendix F, turbidity, which is a function of TSS, was also high in groundwater samples, with an arithmetic average of 85 nephelometric turbidity units (NTU) and a range of less than 0.1 to 790 NTU (see Table F3-28 in Appendix F). Typically, groundwater is less than 5 NTU, with reported values up to 19 NTU in natural groundwater (Wilde and Radtke 1998). These results indicate that surrounding sediment and/or native materials were drawn into the samples during collection from the temporary wellpoints.

4.9.2.2 Vertical Distribution

The vertical distribution of salinity throughout the Study Area was evaluated through a series of longitudinal profiles, depth profiles, and box plot depth profiles by reach (see Figures 4-230 through 4-232) that compare salinity values over depth, from surface water to porewater to groundwater. Longitudinal profiles and box plot depth profiles include the Phase 2 samples collected from shallow porewater (taken from the top 30 cm [12 inches] of the sediment), mid-depth porewater, and groundwater. Depth profiles show the vertical distribution of salinity at the 17 Phase 2 groundwater program locations where shallow porewater, mid-depth porewater, and groundwater samples were collected. Salinity was not analyzed in the shallow porewater samples at these locations, so salinity concentrations measured at the nearest triad station are plotted. Additionally, surface water salinity measured within 3 feet of the mudline at Phase 1 and Phase 2 dry weather surface water stations were averaged for each reach of the Study Area and are shown on each depth profile.

As shown in Figures 4-231a through 4-232, salinity in dry weather surface water ranges from 20 to 25 psu (see also Sections 4.7.2.1 and 4.8.2.1). Salinity values at all depths, including surface porewater, mid-depth porewater, and groundwater, between CM 0 and 1.3, are similar to surface water salinity; these data are consistent with relatively low rates of positive (upward) groundwater seepage, and in some cases, slight negative (downward) seepage in CM 0 to 1.3 (see Figures 4-231a through 4-232 and Section 5.2 of Appendix F for discussion of seepage directions). Beginning around CM 1.4, salinity in groundwater declines to less than 10 psu going upstream in all but two samples near CM 1.7 and CM 1.8 (see Figure 4-230). Salinity upstream of CM 1.3 in surface porewater and mid-depth porewater remains closer to surface water salinity than the groundwater, but does show a slight decrease moving upstream, consistent with the groundwater salinity pattern. The salinity pattern upstream of CM 1.3 is generally indicative of upward groundwater flow in these areas.

4.9.3 Distribution of Contaminants

Similar to other media, this section describes the nature and extent of contamination in sampled groundwater beneath the Study Area based on the dissolved phase concentrations of contaminants, with a focus primarily on spatial patterns. Discussions regarding proximity to upland sites is not included, because it would be speculative. Contaminants in the Study Area cannot necessarily be attributed to proximate upland site(s) or source(s), including point sources, due to the complex patterns of development and use of upland sites and the complex history of sediment deposition and transport in the Study Area, as well as dredging and modifications to the course of the creek over time and a lack of upland data for many sites. In addition, contaminant concentrations in groundwater that discharges to the Study Area may represent contributions from any number of potential upland sources within a groundwater contribution area, adding further to the complexity of historical and ongoing contaminant migration to the Study Area.

The sections that follow present concentrations in groundwater for TPAH (17), TPAH (34), C19-C36, TPCB, Cu, and Pb.[141] As with mid-depth porewater, dissolved phase concentrations of organic constituents in groundwater were estimated based on partitioning calculations using two different methods to bracket the range of uncertainty. As summarized in Section 4.8.3 and discussed in detail in Section 3.7.2.1 of Appendix F, Method 1 used site-specific dry weight partition coefficients with sample-specific measured TSS and DOC, whereas Method 2 used literature-based OC partition coefficients with sample-specific measured TOC and DOC or OC

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[141] 2,3,7,8-TCDD is not evaluated here, because groundwater samples were not analyzed for this chemical.

from collocated native material samples when TOC was non-detect (see Section 3.7.2 of Appendix F for additional detail). Both Methods 1 and 2 were used for TPAH (17), TPAH (34), and TPCB, whereas only Method 2 could be applied for C19-C36, as discussed in Section 4.8.3. Results from both methods are presented in Sections 4.9.3.1 and 4.9.3.2; in most cases, estimated dissolved phase groundwater concentrations derived from Method 2 are higher than concentrations derived from Method 1. Method 2 may be more appropriate for estimating dissolved-phase concentrations in groundwater, because of the different sorbent properties between the native material and Study Area sediment. However, because there is uncertainty in both methods, both methods were evaluated and are presented here (see Section 3.7.2.1 of Appendix F for more detail). As with mid-depth porewater, concentrations for Cu and Pb described in this section are based on dissolved results, which were measured analytically using field-filtered samples.

4.9.3.1 TPAH (17)

4.9.3.1.1 Spatial Distribution

Estimated dissolved phase TPAH (17) concentrations in groundwater beneath each reach of the Study Area are summarized in Tables 4-89a and 4-89b. A plan view map and longitudinal profile showing the data are provided in Figures 4-233a through 4-234b, respectively.

The ranges, arithmetic averages, and medians for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. TPAH (17) concentrations range from 0.53 to 1,500 µg/L for Method 1 and from 1.0 to 2,000 µg/L for Method 2 (arithmetic averages of 340 and 460 µg/L, respectively; medians of 30 and 43 µg/L, respectively). Concentrations in this reach increase with distance upstream, with the highest concentrations observed near the confluence with Dutch Kills.
- **CM 1 – 2**. TPAH (17) concentrations range from 0.14 to 9.9 µg/L for Method 1 and from 0.19 to 49 µg/L for Method 2 (arithmetic averages of 3.2 and 13 µg/L, respectively; medians of 1.3 and 9.0 µg/L, respectively). Concentrations are generally lower than concentrations in CM 0 – 1 and CM 2+.
- **CM 2+**. TPAH (17) concentrations range from 0.16 to 8,300 µg/L for Method 1 and from 0.72 to 9,000 µg/L for Method 2 (arithmetic averages of 750 and 1,400 µg/L, respectively;

medians of 6.5 and 21 µg/L, respectively). The highest concentrations in the Study Area are observed in CM 2+. Concentrations vary by four orders of magnitude.

- **English Kills**. TPAH (17) concentrations range from 0.11 to 1,300 µg/L for Method 1 and from 0.21 to 4,400 µg/L for Method 2 (arithmetic averages of 130 and 410 µg/L, respectively; medians of 1.9 and 4.0 µg/L, respectively). Concentrations from two samples in the lower 0.5 mile of English Kills are among the highest in the Study Area and are more than an order of magnitude higher than the remaining samples in this tributary. Concentrations decline with distance toward the head of the tributary and toward its mouth.
- **East Branch and Maspeth Creek**. Concentrations in these two tributaries are relatively low and do not exhibit a gradient. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **East Branch**. TPAH (17) concentrations range from 0.23 to 5.6 µg/L for Method 1 and 0.51 to 20 µg/L for Method 2 (arithmetic averages of 2.4 and 7.6 µg/L, respectively; medians of 1.6 and 5.0 µg/L, respectively).
  - **Maspeth Creek**. TPAH (17) concentrations range from 0.57 to 8.8 µg/L for Method 1 and from 3.9 to 21 µg/L for Method 2 (arithmetic averages of 4.0 and 11 µg/L, respectively; medians of 2.6 and 9.1 µg/L, respectively).
- **Dutch Kills and Whale Creek.** Concentrations are similar to those in the nearby main stem in these two tributaries. Concentrations in the middle of Dutch Kills are lower than those at either end. Ranges, arithmetic averages, and medians of concentrations are as follows:
  - **Dutch Kills**. TPAH (17) concentrations range from 0.11 to 36 µg/L for Method 1 and from 2.0 to 46 µg/L for Method 2 (arithmetic averages of 9.4 and 17 µg/L, respectively; medians of 1.4 and 3.2 µg/L, respectively).
  - **Whale Creek**. TPAH (17) concentrations range from 0.46 to 7.6 µg/L for Method 1 and from 0.76 to 12 µg/L for Method 2 (arithmetic averages of 3.0 µg/L and 7.1 µg/L, respectively; medians of 1.0 and 8.5 µg/L, respectively).

As discussed in Section 6 of Appendix F, estimated dissolved phase TPAH (17) concentrations at wells NC075GW (estimated dissolved phase TPAH (17) of 8,300 µg/L for Method 1 and 9,000 µg/L for Method 2), EK093GW (estimated dissolved phase TPAH (17) of 1,300 µg/L for

Method 1 and 4,400 µg/L for Method 2), and NC296GW (estimated dissolved phase TPAH (17) of 360 µg/L for Method 1 and 440 µg/L for Method 2) may be above the actual dissolved concentrations in these samples, due to potential interferences from NAPL. NAPL and/or sheen were confirmed by shake test in native material samples at each of these locations during the monitoring well installation. The cores at two of these three locations (NC075 and NC296 in CM 2+) are Category 2/3 NAPL cores, while the core at the other location (EK093 in English Kills) is a Category 1B NAPL core. Details of visual NAPL observations and shake test results at these locations can be found in Sections 5.4.2 and 5.4.3 of Appendix C. Therefore, the dissolved phase TPAH (17) concentrations for these three samples are considered conservatively high estimates.

4.9.3.1.2 Vertical Distribution

Longitudinal profiles, depth profiles, and box plot depth profiles comparing dissolved phase TPAH (17) concentrations with depth in porewater and groundwater throughout the Study Area are presented in Figures 4-235a through 4-237b, and Tables 4-90a and 4-90b.[142] The assessment of vertical distribution of concentrations focuses mainly on shallow porewater and groundwater; for each of these depths, at least 64 data points are available.[143] Where collocated, mid-depth porewater data exist, and noteworthy results are also discussed for these data. Notable patterns in the data are as follows:

- **CM 0 – 1, CM 1 – 2, and CM 2+**. The arithmetic average TPAH (17) concentrations in groundwater in these reaches are one to three orders of magnitude higher than those in shallow porewater. The collocated, mid-depth porewater concentration is higher than the shallow porewater concentration at all six collocated locations and higher than (or

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[142] Tables 4-90a and 4-90b show results for both dissolved phase porewater and groundwater, as well as the corresponding bulk phase (i.e., sediment and native material) results, for the 17 locations where samples were collected from all four depths (i.e., 0 to 15 cm [0 to 6 inches], 15 to 30 cm [6 to 12 inches], and mid-depth porewater and groundwater) as part of the groundwater investigation. This table allows an assessment of the relative changes in concentration with depth for both sorbed and dissolved phase concentrations. Two versions of the table are presented: one for Method 1 (Table 4-90a) and one for Method 2 (Table 4-90b).

[143] Although these datasets provide a robust means of evaluating vertical differences, they are based on different sampling methods (i.e., passive sampling for shallow porewater, which represents an average over the course of sample collection and may be affected by tidal exchange [see Section 4.8.1], versus temporary wellpoints for groundwater, which represent a single point in time and were affected by elevated TSS and turbidity [see Section 4.9.1]). Although these sampling method differences do not preclude making comparisons between the datasets, they should be acknowledged.

equal to) the groundwater concentration at five of the six collocated, mid-depth sampling locations between CM 1.3 and CM 2.4 (see Tables 4-90a and 4-90b).

- **English Kills**. TPAH (17) concentrations in groundwater are generally lower than (Method 1) or equal to (Method 2) concentrations in shallow porewater, but due to two locations with elevated groundwater concentrations (EK093GW and EK094GW), the arithmetic average groundwater concentration is higher than that for shallow porewater. The collocated, mid-depth porewater concentration is similar to the groundwater concentration at one of two collocated locations (EK093GW) and less than both shallow porewater and groundwater at the other collocated location.
- **East Branch, Maspeth Creek, Dutch Kills, and Whale Creek**. Arithmetic average TPAH (17) concentrations in groundwater in these reaches are generally similar to, or slightly higher than (within a factor of 2), those in shallow porewater. Collocated, mid-depth concentrations in East Branch and Dutch Kills are similar to, but lower than, those in groundwater. Mid-depth porewater samples were not collected from Maspeth Creek and Whale Creek.

Additional discussion of groundwater and porewater concentrations, in the context of fate and transport processes, is provided in Sections 6.4.5 and 6.5.

4.9.3.2 TPAH (34)

4.9.3.2.1 Spatial Distribution

Dissolved TPAH (34) concentrations in groundwater beneath each reach of the Study Area are summarized in Tables 4-91a and 4-91b; a plan view map and longitudinal profile showing the data are provided in Figures 4-238a through 4-239b, respectively.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. TPAH (34) concentrations range from 1.1 to 1,600 µg/L for Method 1 and from 1.8 to 2,100 µg/L for Method 2 (arithmetic averages of 370 and 500 µg/L, respectively; medians of 36 and 54 µg/L, respectively). Concentrations in this reach increase with distance upstream, with the highest concentrations observed near the confluence with Dutch Kills.

- **CM 1 – 2**. TPAH (34) concentrations range from 0.44 to 16 µg/L for Method 1 and from 0.45 to 260 µg/L for Method 2 (arithmetic averages of 7.7 and 40 µg/L, respectively; medians of 5.1 and 15 µg/L, respectively). The highest concentrations in this reach are lower than the highest concentrations in CM 0 – 1 and CM 2+.
- **CM 2+**. TPAH (34) concentrations range from 0.21 to 9,000 µg/L for Method 1 and from 1.1 to 9,600 µg/L for Method 2 (arithmetic averages of 840 and 1,600 µg/L, respectively; medians of 15 and 42 µg/L, respectively). Longitudinal profiles for both Method 1 and Method 2 show a wide degree of variation in TPAH (34) concentrations in this reach (by four orders of magnitude), with the highest concentrations measured in the Study Area being collected from this reach.
- **English Kills**. TPAH (34) concentrations range from 0.32 to 1,500 µg/L for Method 1 and from 0.55 to 4,900 µg/L for Method 2 (arithmetic averages of 160 and 460 µg/L, respectively; medians of 6.0 and 10 µg/L, respectively). Concentrations from two samples in the lower 0.5 mile of English Kills are among the highest in the Study Area and are more than an order of magnitude higher than the remaining samples in this tributary. Concentrations decline with distance toward the head of the tributary and toward its mouth.
- **East Branch**. TPAH (34) concentrations range from 0.59 to 12 µg/L for Method 1 and 1.5 to 65 µg/L for Method 2 (arithmetic averages of 6.6 and 18 µg/L, respectively; medians of 7.3 and 12 µg/L, respectively).
- **Maspeth Creek**. TPAH (34) concentrations range from 1.0 to 24 µg/L for Method 1 and from 6.1 to 49 µg/L for Method 2 (arithmetic averages of 10 and 24 µg/L, respectively; medians of 5.8 and 18 µg/L, respectively).
- **Dutch Kills**. TPAH (34) concentrations range from 0.33 to 49 µg/L for Method 1 and from 2.9 to 63 µg/L for Method 2 (arithmetic averages of 15 and 28 µg/L, respectively; medians of 5.3 and 8.9 µg/L, respectively). Concentrations do not exhibit a gradient but are highest at the head and mouth of Dutch Kills and lowest in the middle portion of the tributary.
- **Whale Creek**. TPAH (34) concentrations range from 1.7 to 14 µg/L for Method 1 and from 2.0 to 24 µg/L for Method 2 (arithmetic averages of 6.7 µg/L and 15 µg/L, respectively; medians of 4.4 and 19 µg/L, respectively).

As discussed for TPAH (17) in Section 4.9.3.1.1, estimated dissolved phase TPAH (34) concentrations at wells NC075GW, EK093GW, and NC296GW also may be greater than the actual dissolved concentrations in these samples, due to potential interferences from NAPL (based on the analysis presented in Section 6 of Appendix F).

4.9.3.2.2 Vertical Distribution

Longitudinal profiles, depth profiles, and box plot depth profiles comparing dissolved phase TPAH (34) concentrations with depth in porewater and groundwater throughout the Study Area are presented in Figures 4-240a through 4-242b and Tables 4-92a and 4-92b. The assessment of vertical distribution of concentrations focuses mainly on shallow porewater and groundwater. Where collocated, mid-depth porewater data exist, and noteworthy results are also discussed for these data. Notable patterns in the data are as follows:

- **CM 0 – 1, CM 1 – 2, and CM 2+**. The arithmetic average TPAH (34) concentrations in groundwater in these reaches are one to three orders of magnitude higher than those in shallow porewater. The collocated, mid-depth porewater concentration is higher than the shallow porewater concentration at all six collocated locations and higher than (or equal to) the groundwater concentration at five of the six collocated, mid-depth sampling locations between CM 1.3 and CM 2.4 (see Tables 4-92a and 4-92b).
- **English Kills**. TPAH (34) concentrations in groundwater are generally lower than concentrations in shallow porewater, although groundwater concentrations are locally elevated at two locations (EK093GW and EK094GW). The collocated, mid-depth porewater concentration is slightly greater than the groundwater concentration at one of two collocated locations (EK093GW) and less than both shallow porewater and groundwater at the other collocated location.
- **East Branch, Maspeth Creek, Dutch Kills, and Whale Creek**. Arithmetic average TPAH (34) concentrations in groundwater in these reaches are generally similar to, or slightly higher than (within a factor of 2), those in shallow porewater. Collocated, mid-depth concentrations in East Branch and Dutch Kills are similar to, but lower than, those in groundwater. Mid-depth porewater samples were not collected from Maspeth Creek and Whale Creek.

4.9.3.3 C19-C36

4.9.3.3.1 Spatial Distribution

Estimated dissolved phase C19-C36 concentrations in groundwater beneath each reach of the Study Area are summarized in Table 4-93; a plan view map and longitudinal profile showing the data are provided in Figures 4-243 and 4-244, respectively. Dissolved C19-C36 concentration estimates are presented for Method 2 only (because estimates based on Method 1 could not be developed; see Section 4.9.3).

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1**. Only one location had a detected C19-C36 concentration in this reach, with an estimated dissolved phase concentration of 0.0043 mg/L. That value is within the range of the MDLs for the non-detected samples in this reach.[144]
- **CM 1 – 2**. C19-C36 concentrations were only detected at two locations in this reach, with estimated dissolved phase concentrations of 0.012 and 0.032 mg/L. Those values are within the range of the MDLs for the non-detected samples in this reach.
- **CM 2+**. C19-C36 was detected at only one location in this reach, with an estimated dissolved phase concentration of 0.11 mg/L. That value is within the range of the MDLs for the non-detected samples in this reach.
- **English Kills**. C19-C36 concentrations range from 0.0017 to 0.40 mg/L with an arithmetic average of 0.055 mg/L and a median of 0.020 mg/L. While only 3 out of 14 samples had detected C19-C36 concentrations, the estimated dissolved phase concentrations suggest an increase moving upstream in English Kills. Detected concentrations were similar to the range of MDLs for the non-detect samples in this reach.
- **East Branch**. C19-C36 concentrations range from 0.00032 to 0.015 mg/L with an arithmetic average of 0.0040 and a median of 0.0015 mg/L. The results for the three

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[144] There were six groundwater samples collected in CM 0 – 1, but only five of these could be used to estimate dissolved phase contaminant concentrations due to lack of TSS in one sample (see Section 4.8.3 for details on the calculation of estimated dissolved phase contaminant concentrations). C19-C36 was detected for this one sample with a whole-water concentration that is in the range of the remaining CM 0 – 1 whole-water C19-C36 concentrations.

detected samples in this reach are within the range of the MDLs for the five non-detected samples in this reach.

- **Maspeth Creek, Dutch Kills, and Whale Creek**. C19-C36 was detected in one sample from each of these tributaries: 0.048 mg/L in Maspeth Creek, 0.19 mg/L in Dutch Kills (which is the highest detected value of the Study Area), and 0.0010 mg/L in Whale Creek. These detected results are higher than the MDLs from the other non-detect samples in each of these reaches but are within the range of MDLs for the Study Area as a whole.

Based on the low FoD for C19-C36 in groundwater (20% across the 64 Study Area samples), coupled with the estimated concentrations for the detected samples being similar to the range of MDLs for the non-detect samples, and considering the uncertainty in the dissolved phase estimation method, these results likely indicate that C19-C36 is generally not present, or is at low to undetectable concentrations, in groundwater.

4.9.3.3.2 Vertical Distribution

Longitudinal profiles, depth profiles, and box plot depth profiles comparing dissolved phase C19-C36 concentrations with depth in porewater and groundwater throughout the Study Area are presented in Figures 4-245 through 4-247 and Table 4-94. Due to the low FoD for C19-C36 in groundwater and mid-depth porewater and the similarity between estimated dissolved phase concentrations for detected samples and MDLs for non-detected samples, as well as the lack of C19-C36 measurements in shallow porewater, vertical distribution patterns, if present, are not able to be ascertained.

4.9.3.4 TPCB

4.9.3.4.1 Spatial Distribution

Estimated dissolved phase TPCB concentrations in groundwater in each reach of the Study Area are summarized in Tables 4-95a and 4-95b; a plan view map and a longitudinal profile showing the data are provided in Figures 4-248a through 4-249b.

The ranges, arithmetic averages, and medians for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1, CM 1 – 2, and CM 2+**. Concentrations in these reaches are generally similar to one another, and relatively low (i.e., less than 10 ng/L in all but two samples; one each in CM 0 – 1 and CM 1 – 2) for Method 1. Results for Method 2 are more variable and have no discernable trend. In addition, maximum concentrations estimated for Method 2 are 3 to 24 times higher than the corresponding Method 1 results.
    - **CM 0 – 1**. TPCB concentrations range from 1.3 to 15 ng/L for Method 1 and from 0.89 to 64 ng/L for Method 2 (arithmetic averages of 5.9 and 18 ng/L, respectively; medians of 3.6 and 8.6 ng/L, respectively).
    - **CM 1 – 2**. TPCB concentrations range from 0.12 to 18 ng/L for Method 1 and from 0.66 to 440 ng/L for Method 2 (arithmetic averages of 4.1 and 50 ng/L, respectively; medians of 1.4 and 5.0 ng/L, respectively).
    - **CM 2+**. TPCB concentrations range from 0.051 to 5.8 ng/L for Method 1 and from 0.91 to 18 ng/L for Method 2 (arithmetic averages of 2.4 and 4.7 ng/L, respectively; medians of 2.4 and 2.9 ng/L, respectively).
- **English Kills**. TPCB concentrations range from 3.2 to 480 ng/L for Method 1 and from 11 to 2,700 ng/L for Method 2 (arithmetic averages of 71 and 460 ng/L, respectively; medians of 19 and 75 ng/L, respectively). The highest concentrations in the Study Area are observed in English Kills. Concentrations in the middle portion are higher than those at either end.
- **East Branch**. TPCB concentrations range from 2.1 to 71 ng/L for Method 1 and from 5.5 to 710 ng/L for Method 2 (arithmetic averages of 19 and 120 ng/L, respectively; medians of 10 and 18 ng/L, respectively). Concentrations are higher than those in the adjacent main stem, but they are similar to those at the downstream end of English Kills.
- **Maspeth Creek**. TPCB concentrations range from 0.074 to 21 ng/L for Method 1 and from 0.52 to 130 ng/L for Method 2 (arithmetic averages of 7.3 and 43 ng/L, respectively; medians of 0.42 and 2.4 ng/L, respectively). Concentrations are similar to those in the main stem.
- **Dutch Kills**. TPCB concentrations range from 0.75 to 63 ng/L for Method 1 and from 5.3 to 570 ng/L for Method 2 (arithmetic averages of 17 and 99 ng/L, respectively; medians of 6.5 and 11 ng/L, respectively). Concentrations are similar to those at the

confluence of the main stem, but they increase upstream toward the head of the tributary.

- **Whale Creek**. TPCB concentrations range from 1.5 to 43 ng/L for Method 1 and from 1.6 to 320 ng/L for Method 2 (arithmetic averages of 16 and 110 ng/L, respectively; medians of 3.4 and 2.9 ng/L, respectively). Concentrations are similar to those at the confluence of the main stem, with a higher concentration sample nearest the head of the tributary.

4.9.3.4.2 Vertical Distribution

Longitudinal profiles, depth profiles, and box plot depth profiles comparing estimated dissolved phase TPCB concentrations with depth in porewater and groundwater throughout the Study Area are presented in Figures 4-250a through 4-252b and Tables 4-96a and 4-96b (with results shown for both Method 1 and Method 2). The assessment of vertical distribution of concentrations focuses mainly on shallow porewater and groundwater; for each of these depths, at least 64 data points are available. Where collocated, mid-depth porewater data exist, noteworthy results are also discussed. In general, TPCB concentrations in the groundwater are similar to, or lower than, the mid-depth porewater for most parts of the Study Area.

- **CM 0 – 1 and CM 1 – 2**. The arithmetic average TPCB concentrations in shallow porewater are similar to those in groundwater estimated from Method 1 (within a factor of 2), but three to eight times lower than those estimated from Method 2. Mid-depth porewater concentrations are generally similar to those in groundwater; two exceptions are near CM 1.3 (NC277 and NC278), where mid-depth concentrations are higher than those in groundwater (by a factor of approximately 30 to 140).
- **CM 2+**. The arithmetic average TPCB concentration in shallow porewater is higher than that in groundwater (by a factor of approximately 10 to 20, depending on the dissolved phase estimation method). Collocated, mid-depth porewater concentrations are similar to those in groundwater.
- **English Kills**. The arithmetic average TPCB concentration in shallow porewater is approximately two times higher than that in groundwater based on Method 1 and approximately three times lower than that in groundwater based on Method 2. Collocated, mid-depth porewater concentrations are higher than groundwater concentrations (by a factor of 1.9 to 2.3, depending on dissolved phase estimation

method) at one location and lower (by a factor of 0.02 to 0.2, depending on dissolved phase estimation method) at a second location.

- **East Branch**, **Maspeth Creek**, **Dutch Kills, and Whale Creek**. The arithmetic average TPCB concentrations in groundwater are slightly higher than those in shallow porewater (within a factor of 3 based on Method 1 and within a factor of 16 for Method 2), although concentrations are lower at several locations. Collocated, mid-depth porewater concentrations are generally similar to those in groundwater in Dutch Kills, but are higher than the shallow porewater or groundwater concentration at the one location in East Branch. Mid-depth porewater samples were not collected from Maspeth Creek and Whale Creek.

4.9.3.5 Cu

4.9.3.5.1 Spatial Distribution

Dissolved Cu concentrations in groundwater beneath each reach of the Study Area are summarized in Table 4-97; a plan view map and longitudinal profile showing the data are provided in Figures 4-253 and 4-254, respectively.

The range, arithmetic average, and median for each reach and notable patterns in the data are as follows (if patterns are not discussed, it means no notable patterns were observed):

- **CM 0 – 1, CM 1 – 2, and CM 2+**. Dissolved Cu is non-detect in most samples in the main stem, including all samples in CM 0 – 1, approximately half in CM 1 – 2, and all but three in CM 2+. The highest concentration in the Study Area is observed in CM 1 – 2.
  - **CM 0 – 1**. Cu concentrations range from 2.0 to 4.0 µg/L (arithmetic average and median values were not calculated because all samples in this reach were non-detect).
  - **CM 1 – 2**. Cu concentrations range from 1.1 to 160 µg/L (arithmetic average = 20 µg/L; median = 4.0 µg/L).
  - **CM 2+**. Cu concentrations range from 0.90 to 7.8 µg/L (arithmetic average = 3.8 µg/L; median = 4.0 µg/L).
- **Tributaries**. Most measurements are non-detects. Detected concentrations are relatively low—they are less than 10 µg/L in all cases, except one sample at 19 µg/L in East Branch.

- **English Kills**. Cu concentrations range from 2.0 to 8.6 µg/L (arithmetic average = 4.1 µg/L; median = 4.0 µg/L).
- **East Branch**. Cu concentrations range from 0.26 to 19 µg/L (arithmetic average = 3.8 µg/L; median = 2.0 µg/L).
- **Maspeth Creek**. Cu concentrations range from 4.0 to 8.0 µg/L (arithmetic average and median were not calculated because only one sample had a detectable result).
- **Dutch Kills**. Cu concentrations range from 4.0 to 9.1 µg/L (arithmetic average = 5.5 µg/L; median = 4.2 µg/L).
- **Whale Creek**. Cu concentrations range from 2.0 to 4.0 µg/L (arithmetic average and median values were not calculated because all samples in this reach were non-detect).

4.9.3.5.2 Vertical Distribution

Longitudinal profiles, depth profiles, and box plot depth profiles comparing dissolved Cu concentrations with depth throughout the Study Area are presented in Figures 4-255 through 4-257 and Table 4-98. The assessment of vertical distribution of concentrations focuses mainly on shallow porewater and groundwater; for each of these depths, at least 64 data points are available. Where collocated, mid-depth porewater data exist, noteworthy results are also discussed. In general, Cu concentrations are relatively low in shallow porewater, mid-depth porewater, and groundwater (most concentrations are less than 10 µg/L, with many non-detects).

- **CM 0 – 1, East Branch, Maspeth Creek, and Whale Creek**. Cu concentrations are low (with a few exceptions, less than 10 µg/L) in shallow porewater and groundwater in these reaches. One collocated, mid-depth porewater sample at CM 0.1 has a higher Cu detection than either shallow porewater or groundwater.
- **CM 1 – 2**. The arithmetic average Cu concentration in groundwater is higher than that in shallow porewater by a factor of 4. One collocated, mid-depth porewater sample at CM 1.4 has a higher Cu detection than either shallow porewater or groundwater.
- **CM 2+ and English Kills**. The arithmetic average Cu concentrations in shallow porewater are slightly higher than those in groundwater. The collocated, mid-depth Cu concentrations are similar to, or lower than, those detected in shallow porewater and groundwater.

- **Dutch Kills**. Cu concentrations in shallow porewater are similar to those in mid-depth porewater and groundwater.

4.9.3.6 Pb

4.9.3.6.1 Spatial Distribution

Dissolved Pb concentrations in groundwater beneath each reach of the Study Area are summarized in Table 4-99; a plan view map and longitudinal profile showing the data are provided in Figures 4-258 and 4-259, respectively. Dissolved Pb concentrations are mostly non-detect (39 of 64 samples), with detected concentrations being less than 10 µg/L at all but two locations (one in CM 1 – 2 [80 µg/L] and one in East Branch [14 µg/L]). The longitudinal and vertical spatial profiles are also similar to that of Cu.

4.9.3.6.2 Vertical Distribution

Longitudinal profiles, depth profiles, and box plot depth profiles comparing dissolved Pb concentrations with depth throughout the Study Area are presented in Figures 4-260 through 4-262 and Table 4-100. The assessment of vertical distribution of concentrations focuses mainly on shallow porewater and groundwater; however, the majority of paired shallow porewater and groundwater Pb concentrations are non-detect. Collocated mid-depth porewater concentrations are also non-detect in all but two samples, and these samples have the highest Pb concentrations of the collocated samples. In general, Pb concentrations are relatively low in shallow porewater, mid-depth porewater, and groundwater (most concentrations are less than 10 µg/L, with many non-detects).

4.10 Tissue

4.10.1 Tissue Dataset

The tissue dataset includes results for fish, crab, caged bivalves, and polychaetes that were collected during three sampling programs. These programs (the fish and crab community survey and tissue sampling program; the caged bivalve study; and the surface sediment bioaccumulation study) were conducted to support exposure assessments in the BHHRA and BERA (see Appendices H and I). Additional details concerning the tissue data are discussed in the risk assessments, which are summarized in Section 7 and included in Appendices H and I. A brief description of the sampling programs (summarized in Table 4-101) and discussion of chemical concentrations in tissue are provided in the subsections that follow. Unless otherwise noted, tissue concentrations are reported on a wet weight basis.

4.10.2 Sample Types

4.10.2.1 Fish and Crab

Fish and crab sampling occurred in six zones (FSZs 1, 2, 3, 4a, 4b, and 5) within the Study Area (see Figures 2-19a through 2-19d) and also in the four Phase 2 reference areas (see Figure 2-2). Sampling conducted to support the BHHRA (see Section 3.1.2.2.3 of Appendix H) was focused in FSZs 1 and 3, because most of the fishing and crabbing activities observed (although infrequent) were in these zones. Species and tissue types varied between the BHHRA and BERA programs (see Appendices H and I, respectively, as well as Section 4.1.3.6). The species and tissue composites[145] from the Study Area and Phase 2 reference areas are summarized for TPAH (17),[146] TPCB, 2,3,7,8-TCDD, Cu, Pb, and dieldrin in the sections that follow.

4.10.2.2 Bivalves

A caged bivalve study using the ribbed mussel (*Geukensia demissa*) was conducted to support the assessment of risks to epibenthic invertebrates in the Study Area. The bivalve tissue data were also used as part of the dietary uptake for the fish risk analyses and the wildlife risk analyses (see Section 4.2.5.2 of Appendix I). Mussels were placed in cages and deployed at 10 discrete stations in the Study Area, approximately 2 feet above mudline (see Figures 2-19a through 2-19d). After 60 days of exposure, the bivalves were harvested, and composite samples were created using soft tissue. Bivalves were not deployed in the Phase 2 reference areas.

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[145] Composite samples comprising individual fish or crab (of the same species) were collected from within the same FSZ. Therefore, composite sample results represent overall TPAH (17), TPCB, and Cu concentrations for the entire zone in which they were collected.

[146] The analytical method used for tissue analysis of PAHs included the compounds in TPAH (17) but not TPAH (34). Also, C19-C36 was not analyzed in tissue samples.

4.10.2.3 Benthic Invertebrates

Laboratory-based bioaccumulation tests were performed to evaluate the accumulation of contaminants in benthic invertebrates due to exposure to contaminated surface sediment. The tests were performed by exposing the polychaete bristle worm (*Nereis virens*, formerly known as *Neanthes virens*) to surface sediment collected from 13 of the 36 triad stations in the Study Area (see Figures 2-19a through 2-19d). The results were used in the BERA (see Section 4.2.4.3.4 of Appendix I) to evaluate the potential for adverse effects in benthic invertebrates and to estimate dietary doses to receptors that consume invertebrates as part of their diets. No bioaccumulation testing was performed for the Phase 2 reference areas.

4.10.3 Chemical Concentrations in Tissue

4.10.3.1 TPAH (17)

A box plot summarizing the distribution of TPAH (17) in fish and crab from the Study Area and the Phase 2 reference areas is presented in Figure 4-263 on both log and linear scales. Longitudinal profiles of TPAH (17) concentrations in fish, crab, bivalves, and polychaetes by tissue type are shown in Figures 4-264 through 4-270; fish and crab are presented on linear scales, while caged bivalves and polychaetes are presented on log scales. Summary statistics are presented in Tables 4-102a through 4-102i. For fish and crab, FoD of TPAH (17) ranges from 70% to 100% in the Study Area, and 35% to 90% in the Phase 2 reference areas. PAHs were detected in all caged bivalve and polychaete samples.

Study Area TPAH (17) concentrations are greater than reference area concentrations for all species (Figure 4-263 shows box plots that provide the median, 10th, 25th, 75th, and 90th percentiles of the distributions). Among all species, TPAH (17) concentrations are highest in caged bivalves and polychaetes. Notable observations by species are as follows (concentration values presented are ranges across the full Study Area, with an indication of the zones in which the minimum and maximum concentrations occurred):

- **Striped bass**. TPAH (17) concentrations in fillet samples from the Study Area range from 0.0054 mg/kg in FSZ 1 to 0.059 mg/kg in FSZ 3. TPAH (17) concentrations in whole-body samples range from 0.030 mg/kg in FSZs 1 and 3 to 0.18 mg/kg in FSZ 4a. Overall, Study Area TPAH (17) concentrations are lowest in FSZ 1. There is an increase in TPAH (17) concentrations in tissue with distance upstream. Concentrations in the Study Area are higher than concentrations in the reference areas.

- **White perch**. TPAH (17) concentrations in fillet samples from the Study Area range from 0.023 mg/kg in FSZ 4b to 0.053 mg/kg in FSZ 3. Concentrations in FSZs 4b and 5 are comparable to the data collected in Westchester Creek. Data are more limited than for the other species.
- **Atlantic menhaden**. TPAH (17) concentrations in whole-body samples from the Study Area range from 0.066 mg/kg in FSZ 4b to 0.29 mg/kg in FSZ 5 and are higher in Atlantic menhaden than for any other fish. The lowest concentrations overall were measured in FSZs 1 and 3 (in CM 0 – 2). Concentrations from all Study Area zones lie within the range of concentrations measured from the Westchester Creek reference area.
- **Mummichog**. TPAH (17) concentrations in mummichog whole-body samples from the Study Area range from 0.0055 mg/kg in FSZ 3 to 0.19 mg/kg in FSZ 1. In contrast to Atlantic menhaden and striped bass, TPAH (17) concentrations generally decrease with distance upstream from the mouth (except for two high values in FSZ 4b). Concentrations throughout the Study Area lie within the range observed in Westchester Creek.
- **Blue crab**. Whole-body TPAH (17) concentrations in the Study Area range from 0.0048 mg/kg in FSZ 2 to 0.085 mg/kg in FSZ 3 and exhibit no spatial gradient. In contrast, there is a general decrease in concentration for the hepatopancreas and muscle (these tissues are combined, per the BHHRA [see Section 3.2.4.4 of Appendix H]) from the mouth of Newtown Creek toward the tributaries at the head of the creek. TPAH (17) concentrations for hepatopancreas and muscle from the Study Area range from 0.046 mg/kg in FSZ 5 to 0.13 mg/kg in FSZ 1. However, it should be noted that the whole-body dataset is larger (i.e., 24 whole-body samples) and has multiple samples from each FSZ (i.e., 4 whole-body samples per FSZ) compared to the hepatopancreas and muscle dataset (i.e., 10 hepatopancreas and muscle samples total, with only 1 sample from four of the six FSZs). Therefore, the lack of a spatial pattern in the whole-body tissue is a more robust observation. TPAH (17) concentrations in both blue crab tissue types are higher in the Study Area than in the reference areas.

- **Caged bivalves**. TPAH (17) concentrations range from 0.085 mg/kg in CM 0 – 1 to 0.75 mg/kg in English Kills, are higher than concentrations in fish and crab, and increase slightly from the mouth of Newtown Creek upstream.
- **Polychaetes**. TPAH (17) concentrations range from 0.026 mg/kg in CM 1 – 2 to 0.44 mg/kg in English Kills and are also higher than concentrations in fish and crab. The next highest concentrations are observed in CM 2+.

4.10.3.2 TPCB

A box plot summarizing the distribution of TPCB in fish and crab from the Study Area and the Phase 2 reference areas is presented in Figure 4-271 on both log and linear scales. Longitudinal profiles of TPCB in fish, crab, bivalves, and polychaetes by tissue type are shown in Figures 4-272 through 4-278; fish and crab are presented on linear scales while caged bivalves and polychaetes are presented on log scales. Summary statistics are presented in Tables 4-103a through 4-103i. PCBs were detected in all tissue samples in the Study Area and all four of the Phase 2 reference areas.

Study Area TPCB concentrations are greater than reference area concentrations for all species (see Figure 4-271). Striped bass whole-body tissue samples from the Study Area have the highest concentrations of TPCB compared with the other species and tissue types. Arithmetic average concentrations in blue crab are similar to those in fish and are higher in hepatopancreas and muscle tissue than in whole-body samples. Notable observations by species are as follows (concentration values presented are ranges across the full Study Area, with an indication of the zones in which the minimum and maximum concentrations occurred):

- **Striped bass**. TPCB concentrations in fillets are generally similar among the FSZs of the Study Area. TPCB concentrations in the Study Area range from 0.16 mg/kg in FSZ 4a to 1.1 mg/kg in FSZ 5. Whole-body TPCB concentrations measured in the Study Area zones overlap considerably (ranging from 0.65 to 3.9 mg/kg in FSZs 2 and 5, respectively), although a general increase is observed in the data, from the mouth of Newtown Creek (FSZ 1) to English Kills and East Branch (FSZ 5). Concentrations in the Study Area are higher than concentrations in the reference areas.

- **White perch**. There are too few data to evaluate spatial patterns within the Study Area, although TPCB in some samples from the Study Area exceed concentrations in the only reference area for which data are available, Westchester Creek. TPCB concentrations in white perch fillet tissue range from 0.095 mg/kg in FSZ 4b to 0.55 mg/kg in FSZ 3.
- **Atlantic menhaden**. Study Area TPCB concentrations in whole-body tissue range from 0.19 mg/kg in FSZ 4b to 3.2 mg/kg in FSZ 3 and generally increase from the mouth of Newtown Creek (FSZ 1) upstream to English Kills and East Branch (FSZ 5). Study Area concentrations generally exceed reference area concentrations.
- **Mummichog**. TPCB concentrations are similar among sampling zones, with the exception of two samples from FSZ 2 with higher concentrations. Both samples are from Dutch Kills, and the concentrations for these samples are 16 and 10 mg/kg. These concentrations are likely elevated due to the locally elevated concentrations in surface sediment (see Section 6.6). TPCB concentrations in the Study Area range from 0.16 mg/kg in FSZ 4b to 16 mg/kg in FSZ 2. Concentrations in the Study Area are generally higher than concentrations in the reference areas.
- **Blue crab**. There is no apparent spatial gradient in TPCB concentrations in blue crab muscle and hepatopancreas; TPCB concentrations in the Study Area range from 0.45 mg/kg in FSZ 3 to 0.69 mg/kg in FSZ 1. Whole-body tissue TPCB concentrations range from 0.080 mg/kg in FSZ 1 to 0.42 mg/kg in FSZ 2. The highest TPCB concentration is in two whole-body samples from FSZ 2 (Dutch Kills), similar to mummichog. TPCB concentrations in the Study Area are higher than the reference areas.
- **Caged bivalves**. TPCB concentrations increase from the mouth of Newtown Creek to the upstream reaches and range from 0.011 mg/kg in CM 0 – 1 to 0.22 mg/kg in English Kills.
- **Polychaetes**. TPCB concentrations range from 0.020 mg/kg in East Branch and Maspeth Creek to 1.2 mg/kg in English Kills and generally increase from the mouth of the creek to the upstream reaches, although concentrations in samples from Maspeth Creek and East Branch are comparable to the lower portion of Newtown Creek. Samples from Dutch Kills, English Kills, and CM 2+ exhibit higher concentrations compared to the main stem samples. This variability is likely due to local variability in surface sediment concentrations (see Section 6.6).

4.10.3.3 2,3,7,8-TCDD

A box plot summarizing the distribution of 2,3,7,8-TCDD in fish and crab from the Study Area and the Phase 2 reference areas is presented in Figure 4-279 on both log and linear scales. Longitudinal profiles of 2,3,7,8-TCDD in fish, crab, bivalves, and polychaetes by tissue type are shown in Figures 4-280 through 4-286; fish and crab are presented on linear scales while caged bivalves and polychaetes are presented on log scales. Summary statistics are presented in Tables 4-104a through 4-104i. For fish and crab, FoD of 2,3,7,8-TCDD ranges from 25% to 100% in the Study Area, and 40% to 100% in the reference areas. For bivalves, FoD for 2,3,7,8-TCDD ranges from 0% to 100% in the Study Area, and for polychaetes, FoD ranges from 40% to 100% in the Study Area (there are no reference area tissue data for bivalves and polychaetes).

Median tissue concentrations of 2,3,7,8-TCDD for the Study Area are greater than reference area median tissue concentrations for fish and crab (with the exception of white perch), but for Atlantic menhaden, striped bass, and white perch, there is considerable overlap between Study Area and reference area tissue concentrations (see Figure 4-279). 2,3,7,8-TCDD concentrations are highest in Study Area whole-body striped bass and blue crab hepatopancreas and muscle composites. Notable observations by species are as follows (concentration values presented are ranges across the full Study Area, with an indication of the zones in which the minimum and maximum concentrations occurred):

- **Striped bass**. 2,3,7,8-TCDD concentrations in fillets are generally similar among the FSZs of the Study Area, ranging from approximately 0.2 ng/kg in FSZs 1, 3, and 4a to 1.8 ng/kg in FSZ 5. Whole-body 2,3,7,8-TCDD concentrations in the Study Area range from 0.19 ng/kg in FSZ 4a to 4.6 ng/kg in FSZ 3, with one of four composites in FSZ 5 exhibiting a concentration of 8.5 ng/kg. While median Study Area tissue concentrations are slightly higher than for the reference areas, there is considerable overlap in the range of concentrations.
- **White perch**. 2,3,7,8-TCDD concentrations in fillets from the Study Area range from 0.16 to 1.0 ng/kg in FSZ 4b. As for other analytes, there are too few data to evaluate spatial patterns of 2,3,7,8-TCDD in white perch within the Study Area.
- **Atlantic menhaden**. Study Area 2,3,7,8-TCDD concentrations in whole-body Atlantic menhaden show no spatial trend, with concentrations ranging from

approximately 0.3 to 0.4 ng/kg in FSZs 1, 2, 3, 4a, and 4b to approximately 1.0 ng/kg in FSZs 1 and 5 (with concentration greater than 1 ng/kg in FSZ 5). While median concentrations of 2,3,7,8-TCDD in Atlantic menhaden from the Study Area are slightly higher than those from the reference areas, there is considerable overlap in the range of concentrations.

- **Mummichog**. 2,3,7,8-TCDD concentrations in mummichog show no spatial trend, with concentrations ranging from 0.11 ng/kg in FSZ 4b to approximately 0.35 ng/kg in FSZs 1, 2, 4a, 4b, and 5. Depending on the specific Study Area fish zone and the specific reference area, mummichog tissue is similar to or higher in the Study Area compared to the reference areas.
- **Blue crab**. 2,3,7,8-TCDD in Study Area blue crab muscle and hepatopancreas show a general trend of decreasing concentrations from the mouth of Newtown Creek to the upper reaches, with concentrations ranging from 1.2 ng/kg in FSZs 4b and 5 to 6.6 ng/kg in FSZ1. 2,3,7,8-TCDD in Study Area whole-body blue crab show a similar trend, with maximum concentrations ranging from to 0.24 ng/kg in FSZ 4b to 0.76 ng/kg in FSZ1. 2,3,7,8-TCDD concentrations in Study Area blue crab are higher than in blue crab from Head of Bay, Spring Creek, and Gerritsen Creek, but are similar to blue crab from Westchester Creek.
- **Caged bivalves**. 2,3,7,8-TCDD concentrations in caged bivalves show no spatial trend in the Study Area, ranging from approximately 0.1 to 0.2 ng/kg when detected (2,3,7,8-TCDD was not detected in 4 of the 10 composite samples). 2,3,7,8-TCDD was not detected in the control sample (T = 0).
- **Polychaetes**. 2,3,7,8-TCDD concentrations in polychaetes show a general trend for increasing concentrations from the mouth of the creek (approximately 0.08 ng/kg) to the upstream reaches (median of 0.33 ng/kg in English Kills), with lower concentrations in Dutch Kills and East Branch (medians of 0.058 and 0.067 ng/kg, respectively).

4.10.3.4 Cu

A box plot summarizing the distribution of Cu in fish and crab from the Study Area and Phase 2 the reference areas is presented in Figure 4-287 on both log and linear scales. Longitudinal profiles of Cu concentrations in fish, crab, bivalves, and polychaetes by tissue type are shown in Figures 4-288 through 4-294; fish and crab are presented on linear scales while caged bivalves and polychaetes are presented on log scales. Summary statistics are presented in Tables 4-105a through 4-105i. Cu was detected in all tissue samples in the Study Area and all four Phase 2 reference areas, as would be expected because Cu is an essential element that is well regulated by most organisms. Differences observed among species may be attributed to different species-specific Cu requirements.

Comparisons with reference areas are not as clear-cut for Cu as for TPCB and TPAH (17). The similarity between the Study Area and the reference areas and the relatively small range of Cu concentrations observed for all species (notably for polychaetes and caged bivalves) are consistent with the essentiality of Cu and the ability of these organisms to regulate tissue Cu levels. Study Area Cu concentrations are slightly greater than, but generally within the range of, reference area concentrations in blue crab, mummichog, and striped bass whole body (see Figure 4-287). Study Area concentrations are similar to reference area concentrations in Atlantic menhaden, striped bass fillet, and white perch. Blue crab whole-body and muscle/hepatopancreas samples from the Study Area have the highest concentrations of Cu compared with the other species and tissue types, which may reflect Cu storage in the hepatopancreas tissue. Notable observations by species are as follows (concentration values presented are ranges across the full Study Area, with an indication of the zones in which the minimum and maximum concentrations occurred):

- **Striped bass**. Cu concentrations in fillets from the Study Area range from 0.42 mg/kg in FSZ 3 to 0.50 mg/kg in FSZ 5. In whole-body tissue, Cu concentrations range from 0.61 mg/kg in FSZ 1 to 1.6 mg/kg in FSZ 3. Cu concentrations in both fillets and whole body are slightly higher in the Study Area, but generally within the range of the reference areas. Cu concentrations are similar among the sampling zones for both striped bass tissue types.
- **White perch**. No trends are apparent in Cu concentrations among Study Area sampling zones. Cu concentrations in white perch from the Study Area and reference areas are similar; Cu concentrations in the Study Area range from 0.82 mg/kg in FSZ 4b to 1.6 mg/kg (also in FSZ 4b).
- **Atlantic menhaden**. Cu concentrations range from 1.5 mg/kg in FSZs 1 and 2 to 7.1 mg/kg in FSZ 4b and generally increase from the mouth of Newtown Creek to Maspeth Creek (FSZ 4b). Concentrations in the lower portion of the Study Area (i.e., FSZs 1 through 3) are similar to the Jamaica Bay reference areas, while

concentrations in the uppermost portion of the Study Area (i.e., FSZ 5) are similar to the Westchester Creek reference area.

- **Mummichog**. Cu concentrations exhibit no spatial gradient in the Study Area and range from 1.7 mg/kg in FSZ 2 to 3.8 mg/kg in FSZ 3. The highest concentrations measured are from two samples from reference areas (Head of Bay and Gerritsen Creek).
- **Blue crab**. Concentrations of Cu in both tissue types for blue crab are considerably higher (at least an order of magnitude) than other species. Hepatopancreas and muscle Cu concentrations from the Study Area range from 14 mg/kg in FSZ 3 to 24 mg/kg in FSZ 1. Whole-body Cu concentrations in the Study Area range from 9.5 mg/kg in FSZ 2 to 25 mg/kg in FSZs 1 and 4b. No trends are apparent among Study Area sampling zones. Concentrations in the Study Area are somewhat higher, but generally within the range of concentrations in the reference areas.
- **Caged bivalves**. Cu concentrations exhibit little gradient within the Study Area. The overall Cu concentration range is from 0.89 mg/kg in CM 1 – 2 to 2.4 mg/kg in Dutch Kills. Cu in the control sample (T = 0) is within the range of concentrations measured in the Study Area samples.
- **Polychaetes**. Cu increases slightly with distance upstream from the mouth, ranging from 0.97 mg/kg in CM 0 – 1 to 3.9 mg/kg in CM 2+.

4.10.3.5 Pb

A box plot summarizing the distribution of Pb in fish and crab from the Study Area and the Phase 2 reference areas is presented in Figure 4-295 on both log and linear scales. Longitudinal profiles of Pb concentrations in fish, crab, bivalves, and polychaetes by tissue type are shown in Figures 4-296 through 4-302; fish and crab are presented on linear scales while caged bivalves and polychaetes are presented on log scales. Summary statistics are presented in Tables 4-106a through 4-106i. FoD was 100% in all species and tissue types except for striped bass fillets from the reference areas, for which the FoD was 20% to 100%.

Study Area and reference area tissue concentrations of Pb overlap considerably for fish and crab, with Study Area concentrations being higher than reference area concentrations in blue crab hepatopancreas and muscle composites and in striped bass fillets (see Figure 4-295).

Notable observations by species are as follows (concentration values presented are ranges across the full Study Area, with an indication of the zones in which the minimum and maximum concentrations occurred):

- **Striped bass**. Pb concentrations in striped bass fillets from the Study Area are similar among all fish zones, ranging between 0.003 mg/kg in FSZs 1,2, 3, and 5 to 0.008 mg/kg in FSZ 4a, with a higher concentration of 0.043 mg/kg in FSZ 4b. Whole-body striped bass Pb concentrations likewise show no spatial trend in the Study Area, with minimum concentrations between 0.051 and 0.095 mg/kg and maximum concentrations between 0.16 and 0.35 mg/kg. Detected Pb concentrations in striped bass fillets are similar in the Study Area and reference areas, while whole-body striped bass Pb concentrations are slightly higher in the Study Area, but generally within the range of the reference area concentrations.
- **White perch**. Pb concentrations in white perch show no spatial trend in the Study Area and are within the range of data for white perch from Westchester Creek (the only Phase 2 reference area from which white perch were collected). Pb concentrations in Study Area white perch range from 0.007 mg/kg in FSZ 4b to 0.012 mg/kg in FSZ 3.
- **Atlantic menhaden**. Pb concentrations in Atlantic menhaden increase from the mouth of Newtown Creek to the upstream reaches, ranging from 0.59 mg/kg in FSZ 1 to 3.2 mg/kg in FSZ 4a. Concentrations in the lower portion of the Study Area (i.e., FSZs 1 through 3) are similar to the Jamaica Bay reference areas, while concentrations in CM 2+ are similar to the Westchester Creek reference area.
- **Mummichog**. Pb concentrations in Study Area mummichog exhibit no spatial gradient and range from 0.078 mg/kg in FSZ 1 to 0.68 mg/kg in FSZ 4a. Study Area mummichog Pb concentrations are within the range of mummichog reference area concentrations.
- **Blue crab**. Pb concentrations in Study Area blue crab (all tissue types) show no spatial trend. Hepatopancreas and muscle Pb concentrations range from 0.074 mg/kg in FSZ 3 to 0.18 mg/kg in FSZ 4b, and in whole body blue crab range from 0.22 to 1.3 mg/kg in FSZ 2. Pb concentrations in Study Area blue crab (all tissue types) are within the range of reference area blue crab Pb concentrations.

- **Caged bivalves**. Pb concentrations in Study Area caged bivalves show no spatial trend, ranging from 0.17 mg/kg in CM 1 – 2 to 0.41 mg/kg in English Kills. Overall, Study Area Pb concentrations in caged bivalves are higher than the control sample (T = 0).
- **Polychaetes**. Pb concentrations in Study Area polychaetes show little to no spatial trend. Median concentrations range from 0.08 mg/kg in CM 0 – 1 to 0.15 mg/kg in East Branch.

4.10.3.6 Dieldrin

A box plot summarizing the distribution of dieldrin in fish and crab from the Study Area and the Phase 2 reference areas is presented in Figure 4-303 on both log and linear scales. Longitudinal profiles of dieldrin in fish, crab, bivalves, and polychaetes by tissue type are shown in Figures 4-304 through 4-310; fish and crab are presented on linear scales while caged bivalves and polychaetes are presented on log scales. Summary statistics are presented in Tables 4-107a through 4-107i. For fish and crab, FoD of dieldrin ranges from 33% to 100% in the Study Area, and 83% to 100% in the reference areas. For bivalves, FoD of dieldrin ranges from 0% to 100% in the Study Area, and for polychaetes, also ranges from 0% to 100% in the Study Area (there are no reference area tissue data for bivalves and polychaetes).

For the Study Area, dieldrin tissue concentrations tend to be greater than reference area tissue concentrations for fish and crab, but for striped bass and blue crab, there is considerable overlap (see Figure 4-303). Median concentrations of dieldrin in Study Area Atlantic menhaden, mummichog, and whole-body striped bass are similar to one another, and are higher than median concentrations in blue crab. Notable observations by species are as follows (concentration values presented are ranges across the full Study Area, with an indication of the zones in which the minimum and maximum concentrations occurred):

- **Striped bass**. Dieldrin concentrations in fillet and whole-body striped bass show a general increase from the mouth of Newtown Creek to English Kills. Dieldrin concentrations in fillets range from 3.8 μg/kg in FSZ 1 to 12 μg/kg in FSZ 5, and in whole body striped bass, range from 6.2 μg/kg in FSZ 2 to 38 μg/kg in FSZ 5. Dieldrin concentrations in the Study Area fillet and whole-body striped bass are higher than in the reference areas, but there is considerable overlap.

- **White perch**. There are too few data to evaluate the spatial patterns of dieldrin in white perch within the Study Area. Dieldrin concentrations in white perch fillets range from 2.6 to 14 µg/kg in FSZ 4b, and from 1.8 to 7.5 µg/kg in Westchester Creek.
- **Atlantic menhaden**. Study Area dieldrin concentrations in whole-body Atlantic menhaden show a general increase from the mouth of the Newtown Creek to English Kills. Dieldrin tissue concentrations range from 7.2 µg/kg in FSZ 1 to 24 µg/kg in FSZ 5, with one of four composites in FSZ 3 exhibiting a concentration of 59 µg/kg. Study Area concentrations generally exceed reference area concentrations.
- **Mummichog**. The ranges in dieldrin concentrations in mummichog are similar among FSZs 1, 3, and 4b (ranging from 4.3 µg/kg in FSZ 4b to 12 µg/kg in FSZ 3); the ranges are also similar to one another in FSZ 4a and FSZ 5, with higher concentrations that range from 15 µg/kg in FSZ 5 to 28 µg/kg in FSZ 4a. Two mummichog composites from FSZ 2 exhibit higher dieldrin concentrations of 35 µg/kg. Dieldrin concentrations in Study Area mummichog are generally higher than those in reference area mummichog.
- **Blue crab**. Dieldrin concentrations in all tissue types for blue crab from the Study Area show an increase from the mouth of Newtown Creek to English Kills. Dieldrin concentrations in blue crab hepatopancreas and muscle composites range from 3.8 µg/kg in FSZ 1 to 13 µg/kg in FSZ 5, and in whole-body blue crab, range from 0.81 µg/kg in FSZ 1 to 3.0 µg/kg in FSZ 2. Dieldrin concentrations in Study Area blue crab are higher than the reference area blue crab, but there is overlap between them.
- **Caged bivalves**. Dieldrin was only detected in caged bivalves from Maspeth Creek and English Kills at concentrations of 2.5 µg/kg and 4.9 µg/kg, respectively. Dieldrin was not detected in the control sample (T = 0).
- **Polychaetes**. Dieldrin concentrations in polychaetes generally increase from the mouth of Newtown Creek to the upstream reaches of the Study Area, although concentrations in composites from Maspeth Creek and East Branch are comparable to the lower portion of Newtown Creek. Detected concentrations of dieldrin in polychaetes range from approximately 0.5 µg/kg in Whale Creek, Dutch Kills, Maspeth Creek, and East Branch, to 20 µg/kg in English Kills.

4.11 Air

Results from the Phase 1 ambient air sampling program were evaluated to assess the extent to which measured VOC and TPCB concentrations may be attributable to volatilization of these chemicals from the Study Area. The concentrations of VOCs and PCBs measured in the Study Area were compared with concentrations measured at site-specific background locations and with NYSDEC regional background data. In addition, upwind and downwind pairs were evaluated to determine if a significant concentration gradient could be observed as a result of Study Area contributions (i.e., if the downwind concentration is significantly higher than the upwind concentration). Detailed information on this program is presented in the BHHRA (see Section 2 of Attachment C1 of Appendix H) and the *Phase 1 Remedial Investigation Field Program Data Summary Report – Submittal No. 1* (Phase 1 DSR Submittal No. 1; Anchor QEA 2013e).

4.11.1 Background Data Sources

The NYSDEC regional background dataset was downloaded from a statewide NYSDEC online database (Anchor QEA 2013e). Data from the three regional ambient air sampling locations with extensive datasets that are closest to the Newtown Creek Study Area were used (see Figure 4-311) to provide greater context to Study Area detected concentrations. A detailed description of the regional background dataset compilation, refinement, and analysis are presented in the BHHRA (see Section 2.2 of Attachment C1 of Appendix H). These data were not combined with the site-specific background data.

Site-specific background locations were selected to provide adequate spatial distribution for the area surrounding Newtown Creek (see Figure 4-311). The five site-specific background locations were representative of the mixed use industrial/commercial/residential areas surrounding the Study Area and capture conditions upwind and downwind of the Study Area, including areas with vehicular traffic. Two site-specific background locations were on residential streets (BLK303LOT25AR and BLK2660LOT50), one was on a commercial dead-end street (BLK2979LOT24), and two were within commercial areas (BLK69LOT14 and BLK2525LOT1AR).

4.11.2 Ambient Air Results

Table 4-108 presents a summary of the ambient air data, including the arithmetic average and standard deviation (Sd) of concentrations of compounds that were detected in the Study Area; the arithmetic average and Sd of concentrations of VOCs detected in site-specific background samples; the FoD for both of these datasets; and the NYSDEC mean and 95th percentile values.

Spatial profiles for each detected compound are presented in Figures 4-312a through 4-312n. These profiles provide visual comparisons of the Study Area VOC and PCB data with the five background stations, which are represented by the orange symbols to the right of the Study Area data. The 95th percentile and mean NYSDEC concentrations are included in these figures to compare the Study Area results to regional background concentrations.

4.11.2.1 Non-Detect Compounds

Many of the VOC analytes (18 of 44) and the majority of PCB Aroclors (8 of 9) were not detected in either the Study Area samples or the site-specific background samples during the sampling program. These non-detected chemicals are as follows:

- 1,1,2,2-tetrachloroethane
- 1,1,2-trichloroethane
- 1,1-dichloroethane
- 1,1-dichloroethene
- 1,2,4-trichlorobenzene
- 1,2-dichlorobenzene
- 1,2-dichloroethene, trans-
- 1,2-dichloropropane
- 1,2-dichloroethene, trans-
- 1,3-dichloropropene, trans-
- 1,3-dichloropropene, cis-
- Benzyl chloride
- Bromodichloromethane
- Bromoform
- Chlorobenzene
- Ethylene dibromide (1,2-dibromoethane)
- Hexachlorobutadiene
- Vinyl chloride
- Aroclor 1016
- Aroclor 1221
- Aroclor 1232
- Aroclor 1248
- Aroclor 1254
- Aroclor 1260
- Aroclor 1262
- Aroclor 1268

In addition, methyl tertiary butyl ether was detected at only one location (at a low concentration, between the MDL and RL). Because this compound was non-detect at the other locations, it was treated as a non-detect compound.

4.11.2.2 Detected Compounds

Table 4-108 presents a summary of the chemicals for which there was more than one detected result in the Study Area or site-specific background sampling locations. The majority of these 26 compounds have Study Area arithmetic average concentrations within the range of site-specific background. Twenty-three of these compounds were evaluated with a one-sided Wilcoxon's rank-sum test to determine whether statistically significant differences existed between concentrations of these compounds in the Study Area and site-specific background sampling locations. To calculate summary statistics for the Wilcoxon's rank-sum test, both Study Area and site-specific background data were log-transformed to normalize the distribution of the Study Area and site-specific background data (although log transformation of data is not necessary for non-parametric tests such as this, the transformation does not affect the results of this statistical test). Data are summarized as the back-transformed geometric mean and Sd (see Table 4-109). The statistical test was not conducted for cis-1,2-dichloroethene and Aroclor 1242, because there were no detected concentrations of these compounds in the site-specific background samples. The statistical test was not conducted for 1,1,1-tricholorethane because of the low FoD for this compound. In addition, the few detected concentrations for cis-1,2-dichloroethene and 1,1,1-tricholorethane were below the RL.

Results from the one-sided Wilcoxon's rank-sum test for 23 compounds indicated that no compounds were detected in the Study Area at concentrations statistically significantly above those at site-specific background locations ($p > 0.05$; see Table 4-109). The comparison to site-specific background is considered the primary and strongest LOE that the concentrations of chemicals measured in air are within background levels and are not elevated within the Study Area. Based on this analysis, it can be concluded that, overall, detected concentrations in the Study Area are not statistically different (at a 95% confidence level) than concentrations detected at the site-specific background locations.

PCBs (quantified as Aroclor 1242) were detected in 2 of 24 air samples within the Study Area (see Figure 4-313). The location of these samples is not consistent with the location of elevated TPCB concentrations in surface water (see Section 4.7), indicating that volatilization from the Study Area is unlikely to be responsible for these detected concentrations in air. Although one of the detected air measurements is in English Kills, where elevated TPCB are observed in surface water samples (Phase 2 congener data; see Figure 4-163), the second detected air measurement is in Dutch Kills, where water column concentrations are generally not elevated relative to the rest of the Study Area. In addition, other air sampling locations, where PCBs were non-detect, were in the vicinity of locations where elevated surface water concentrations were observed (e.g., CM 2+). In summary, PCBs in ambient air are only rarely detected. Based on the lack of correlation with the spatial patterns of surface water data, these detected concentrations do not appear to be due to volatilization from Newtown Creek, but instead may represent localized upland sources.

Nine upwind and downwind transects along Newtown Creek were evaluated for localized trends in the measured compounds (see Figure 4-313), specifically to determine if higher concentrations would be observed at downwind stations relative to their paired upwind stations. Additional information on the evaluation of upwind and downwind transects and local meteorological data are provided in the Phase 1 DSR Submittal No. 1 (see Anchor QEA 2013e).

A two-sided paired Student's t-test was used to determine whether there were statistically significant differences in concentrations between upwind and downwind samples along Newtown Creek (see Table 4-110). Because most detected results were near the detection limit, the results of the tests were very sensitive to the values selected for non-detects. For example, a detection near the RL could be found to be significantly higher than a non-detect result reported at the MDL, but not a non-detect reported at the RL. To avoid these uncertainties, only transects in which both members of the upwind and downwind pairs were detected were included in the analysis. In addition, compounds with detections in three or fewer sample pairs were excluded from the analysis (i.e., 1,1,1-trichloroethane, cis-1,2-dichloroethene, and Aroclor 1242). Among the remaining 23 compounds, there were no statistically significant differences (at a 95% confidence level) in chemical concentrations between the upwind and downwind stations ($p > 0.05$; see Table 4-110). This indicates that there is no evidence of significant Study Area impacts to air quality for these compounds.

4.12 Collocation Comparison of Contaminants Identified for In-Depth Evaluation

The purpose of this section is to discuss the collocation of contaminants identified for in-depth evaluation in this RI Report to help inform the understanding of contaminant distributions. Although this section focuses on surface sediments, the FS will evaluate remedial alternatives as they pertain to the risk-driving COCs in the media of concern (e.g., surface sediments, subsurface sediments, porewater, groundwater, surface water, and biota).

In the preceding sections, the nature and extent of contamination within the Study Area was discussed for eight different chemicals (or groups of chemicals) for each sampling medium (depending on data availability for that medium). A comparative analysis of these contaminants by class (i.e., hydrocarbons, which include TPAH [17], TPAH [34], and C19-C36; bioaccumulative organics, which include TPCB, 2,3,7,8-TCDD, and dieldrin; and metals, which include Cu and Pb) is provided in the following subsections. Detailed comparisons are presented first for surface sediment (Section 4.12.1), followed by summary-level comparisons for the other media (Section 4.12.2). This document notes that the FS and subsequent remedial decisions may equally consider ongoing inputs, surface sediments, subsurface sediments, surface water, groundwater/porewater, and certain biotic tissues. This evaluation is used to assess similarities in spatial distribution and identify a representative subset for each class that will be used for the evaluation of sources and fate and transport in Sections 5 and 6, respectively, as summarized in Section 4.12.3.

4.12.1 Comparison of Contaminants in Surface Sediment

The subsections that follow present comparisons of the spatial patterns and distribution of surface sediment concentrations for pairs of contaminants within each class in the Study Area. The comparisons of surface sediment concentrations are shown in three types of graphics: longitudinal profiles, box plots by location, and cross plots (see Section 4.1.4 for a general description of these graphical presentation tools). These three types of comparison graphics present the data in varying formats, and there are some differences with each type with respect to how prominently certain features of the datasets are displayed (e.g., spatial patterns, scatter/variability in the data [overall and by reach], and correlation).

The first graphic format used in this comparison provides a longitudinal profile of concentration for all Study Area surface sediment samples. For each pair, the two contaminants are shown with different colors and on primary and secondary y-axes. The y-axis ranges were developed so that the two contaminants are shown with the same number of log cycles, and the scales were set so that the concentrations of the two contaminants in CM 0 – 1 are plotted at approximately the same position (i.e., so the comparison can focus on spatial patterns rather than the concentration values; thus the y-axis ranges are not the same). The rationale behind this scaling is that CM 0 – 1 is heavily influenced by East River solids based on sediment transport modeling (see Section 5 of Appendix G), and concentrations increase moving upstream in most cases (as discussed in Section 4.2.3). Evaluations of ongoing inputs to the Study Area are the subject of a parallel project effort separate from this RI Report. However, using CM 0 – 1 as a "match point" facilitates comparisons of spatial gradients and may also provide a means of understanding relative magnitudes of sources.

The second graphic format compares the data spatially by reach using box plots. This format shows summary statistics by reach, so the degree of scatter from individual data points is not evident. This graphic uses the same approach to the y-axis scales as described for the longitudinal profiles (i.e., same number of log cycles, and use of CM 0 – 1 as a match point) to allow for an evaluation of how the central tendency of the data changes by reach for the two contaminants (which are shown with paired boxes having solid and cross-hatched fills).

The third graphic format presents comparisons of the data using cross plots, with the x-axis and y-axis scales each containing the same number of log cycles. This format allows the degree of correlation (or lack thereof) between the pair of contaminants to be better visualized. Different colors are used for each reach in the Study Area. The plots display concentrations for every individual sample, which allows for visualization of the degree of scatter in the data by reach and Study Area wide.

4.12.1.1 Hydrocarbons

Sections 4.12.1.1.1 and 4.12.1.1.2 present comparisons of the spatial patterns and distribution of various hydrocarbons in surface sediment to those of TPAH (34) and C19-C36, respectively. In addition to TPAH (17), two other hydrocarbon sums are included in these comparisons: TPH and DRO, because as discussed in Section 4.1.2.2.2, USEPA concluded that toxicity to benthic invertebrates is correlated to each of these classes of hydrocarbons, to varying degrees (and recognizing that not all contribute equally).

4.12.1.1.1 Comparisons to TPAH (34)

A comparison of C19-C36 to TPAH (34) is provided in Figures 4-314 through 4-316.[147] C19-C36 surface sediment concentrations in the Study Area generally exhibit a similar spatial pattern to TPAH (34). The lowest C19-C36 and TPAH (34) concentrations are found in CM 0 – 1, and concentrations generally increase going upstream, except for a few low-concentration FS shoreline samples (with the difference between these FS shoreline samples and the remaining samples being more prominent for TPAH [34] than for C19-C36). The highest concentrations for both C19-C36 and TPAH (34) in the main stem are found in CM 2+. Concentrations of both contaminants in the tributaries are generally similar to CM 2+ concentrations. While surface sediment concentrations for both C19-C36 and TPAH (34) increase going upstream in the main stem and are higher in tributaries as compared to CM 0 – 1, there are some differences. First, there is greater variability in C19-C36 concentrations than in TPAH (34) concentrations on smaller scales and in some reaches (e.g., the relative sizes of the interquartile ranges for C19-C36 are larger than those for TPAH [34] in CM 0 – 1 and English Kills; see Figure 4-315). Second, the magnitudes of concentration increases relative to CM 0 – 1 are greater for C19-C36 than for TPAH (34). For example, relative to CM 0 – 1, median values by reach increase by up to 5-fold for TPAH (34) (based on English Kills), whereas the increase is more than 30-fold for C19-C36 (based on East Branch).

A comparison of TPH to TPAH (34) is provided in Figures 4-317 through 4-319, and a comparison of DRO to TPAH (34) is provided in Figures 4-320 through 4-322. TPH and DRO surface sediment concentrations in the Study Area exhibit spatial patterns that are very similar to that of TPAH (34), when considering both the individual samples, as well as the box plots by reach. The only difference observable from these figures is that relative to CM 0 – 1, the increases in concentrations for TPH and DRO are somewhat greater than those for

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[147] Comparisons between TPAH (17) and TPAH (34) are presented and discussed in Section 4.2.5.1, including these same figure formats (see Figures 4-40 through 4-42).

TPAH (34), similar to what was observed in the C19-C36 comparison. The cross plots also illustrate that TPH and DRO exhibit correlation and collocation with TPAH (34). Although the cross plots of the data suggest relatively good correlation (e.g., data for most samples are within a factor of approximately 2 for both chemicals at a given concentration), there is variability in the correlation at the scale of individual samples (especially at the lower ends of the concentration ranges) and some differences by reach (e.g., the scatter of the relationship from CM 0 – 1 is relatively greater than that for East Branch).

4.12.1.1.2 Comparisons to C19-C36

A comparison of TPAH (17) to C19-C36 is provided in Figures 4-323 through 4-325. As with TPAH (34) (see Section 4.12.1.1.1), TPAH (17) surface sediment concentrations in the Study Area generally exhibit a similar spatial pattern to C19-C36, with the lowest concentrations in CM 0 – 1 and concentrations generally increasing going upstream in the main stem and higher concentrations in the tributaries. While surface sediment concentrations for both C19-C36 and TPAH (17) increase going upstream, there is greater variability in C19-C36 concentrations than TPAH (17) concentrations throughout most of the Study Area, and the spatial gradient is larger for C19-C36 than for TPAH (17) (e.g., relative to CM 0 – 1, median values by reach increase by up to approximately 5-fold for TPAH [17] based on CM 2+, whereas the increase is more than 30-fold for C19-C36 based on East Branch; see Figure 4-324). The difference in variability and gradient is also observable in the cross plot, which shows that the majority of TPAH (17) concentrations span approximately two orders of magnitude, while those for C19-C36 span approximately three, and the degree of scatter in the relationship is relatively high, especially at the lower concentration range).

A comparison of TPH to C19-C36 is provided in Figures 4-326 through 4-328, and a comparison of DRO to C19-C36 is provided in Figures 4-329 through 4-331. TPH and DRO surface sediment concentrations in the Study Area exhibit spatial patterns that are similar to C19-C36, when considering both the individual samples, as well as the box plots by reach. Similar to the TPAH (34) and TPAH (17) comparisons to C19-C36, the spatial gradient is steeper for C19-C36 than for TPH and DRO (e.g., relative to CM 0 – 1, median values by reach increase by as much as 10-fold for TPH and DRO [based on English Kills], compared to an increase of more than 30-fold for C19-C36 [based on East Branch]; see Figures 4-327 and 4-330). The cross plots also illustrate that TPH and DRO exhibit correlation and collocation with C19-C36 (see Figures 4-328 and 4-331). Although the data comparisons suggest relatively good correlation, there is variability at the scale of individual samples, especially at the low concentration ranges.

4.12.1.2 Bioaccumulative Organics

A comparison of 2,3,7,8-TCDD to TPCB is provided in Figures 4-332 through 4-334. 2,3,7,8-TCDD surface sediment concentrations in the Study Area generally exhibit a more muted spatial pattern as compared to that of TPCB. The lowest 2,3,7,8-TCDD and TPCB concentrations are found in CM 0 – 1, and concentrations generally increase going upstream in the main stem. Higher concentrations for both 2,3,7,8-TCDD and TPCB are found in CM 2+, although the increase between CM2+ and CM 0 – 1 is greater for TPCB than it is for 2,3,7,8-TCDD (e.g., almost a 10-fold increase in median for TPCB versus approximately 2-fold for 2,3,7,8-TCDD). TPCB and 2,3,7,8-TCDD concentrations in English Kills are similar to their respective CM 2+ concentrations, but Maspeth Creek and East Branch generally have lower concentrations than the adjacent main stem, with this difference being more prominent for 2,3,7,8-TCDD. 2,3,7,8-TCDD concentrations in Dutch Kills are generally similar to concentrations found in the adjacent main stem, while the highest TPCB concentration is found in Dutch Kills. In Whale Creek, TPCB concentrations are similar to those in CM 1 – 2, whereas some of the highest 2,3,7,8-TCDD surface sediment concentrations are found in Whale Creek. While TPCB and 2,3,7,8-TCDD concentrations generally exhibit similar spatial patterns, the spatial gradient is steeper for TPCB than for 2,3,7,8-TCDD. Although the data suggest relatively good correlation spatially, there is variability in the correlation at the scale of individual samples and by reach as illustrated in the cross plot (see Figure 4-334). For example, there are several individual samples in Whale Creek and the lower main stem (i.e., CM 0 – 1 and CM 1 – 2) for which the concentration of 2,3,7,8-TCDD is locally elevated as compared to TPCB. Likewise, the positive relationship between the two contaminants observed in the cross plot appears to differ somewhat by reach, with the values for the main stem and Whale Creek plotting relatively higher than those in English Kills).

A comparison of dieldrin to TPCB is provided in Figures 4-335 through 4-337. Dieldrin and TPCB surface sediment concentrations in the Study Area exhibit similar spatial patterns when considering both the individual samples, as well as the box plots by reach (see Figures 4-335 and 4-336). The lowest dieldrin and TPCB concentrations are found in CM 0 – 1, and concentrations generally increase going upstream in the main stem, with the exception of several Phase 1 non-detect samples for dieldrin (which may be due to the analytical method used, which was changed for Phase 2) and some FS shoreline samples (with the difference between these FS shoreline samples and the remaining samples being more prominent for TPCB than dieldrin). Higher concentrations for both dieldrin and TPCB are found in CM 2+ and the tributaries. On a relative basis, the central tendency and distributions for the two contaminants are more similar in the main stem and lower tributaries (Dutch Kills and Whale Creek), while for the upper tributaries (English Kills, East Branch, and Maspeth Creek), the relative increases for dieldrin are somewhat greater than those for TPCB. For example, relative to CM 0 – 1, median values for TPCB in the three upper tributaries are higher by factors of approximately 2 to 10, whereas those factors are approximately 10 to 30 for dieldrin (see Tables 4-16 and 4-20). The cross plot also illustrates that dieldrin and TPCB exhibit correlation and collocation (see Figure 4-337). Although the cross plot of the data suggests relatively good correlation (e.g., data for most samples are typically within a factor of 2 to 5 at a given concentration for both contaminants), there is variability in the correlation at the scale of individual samples and by reach (with relatively greater scatter for the CM 0 – 2 than for the other reaches).

4.12.1.3 Metals

A comparison of Pb to Cu is provided in Figures 4-338 through 4-340. Pb concentrations in the Study Area surface sediment generally exhibit a similar pattern to Cu. The lowest Pb and Cu concentrations are found in CM 0 – 1, and concentrations generally increase going upstream in the main stem, with the exception of some FS shoreline samples. The highest concentrations for both Pb and Cu are found in CM 2+. Concentrations in the tributaries for both Pb and Cu are higher than the respective concentrations in CM 0 – 1, and by relatively similar amounts (comparing medians and interquartile ranges in Figure 4-339). The one observable difference between the two contaminants is that the relative increase in concentrations in CM 2+ and in CM 1 – 2, to some extent, is greater for Cu than for Pb. For example, compared to CM 0 – 1, the median concentrations for Pb in CM 1 – 2 and CM 2+ are 1.7 and 5.0 times greater, respectively; these factors are 2.2 and 18 for Cu, respectively (see Tables 4-18 and 4-19). The cross plot also illustrates that Cu and Pb exhibit correlation and collocation (see Figure 4-340). Although the cross plot of the data suggests relatively good correlation, there is variability in the correlation at the scale of individual samples and by reach. For example, there are several individual samples in Dutch Kills and English Kills that show relatively higher concentrations of Pb compared to Cu, and the data from CM 2+ as a whole follow a slightly different relationship, with Cu having relatively higher concentrations than Pb, as compared to the remaining reaches.

4.12.2 Comparison of Contaminants in Other Media

Similar to surface sediment, the contaminants evaluated for nature and extent exhibit broadly similar spatial patterns within a given chemical class for other media, as follows:

- As with surface sediment, data from **subsurface sediment and sediment traps** show that the spatial patterns of TPAH (17), TPAH (34), and C19-C36 are broadly similar to one another (see Figures 4-64 through 4-75 and 4-108 through 4-115), the spatial patterns of 2,3,7,8-TCDD are broadly similar to those of TPCB (see Figures 4-76 through 4-83 and 4-117 through 4-121), and the spatial patterns of Pb are broadly similar to those of Cu (see Figures 4-84 through 4-91 and 4-123 through 4-127).[148] Moreover, the fate and transport processes are similar among these chemicals , because these processes are controlled largely by partitioning to particulate matter for each of these chemicals (see Section 6.4.1). There are, however, some differences observed for these media (e.g., concentration gradients differ in some reaches for some chemicals at small spatial scales).
- 2,3,7,8-TCDD was rarely detected in **surface water** during dry weather and wet weather conditions (see Figure 4-184), and Pb was infrequently detected during dry weather conditions (see Figure 4-187). In contrast, TPAH (17), TPAH (34), TPCB, and Cu (see Figures 4-178, 4-182, and 4-185, respectively) were often detected during dry and wet weather conditions (at concentrations within the range of those for samples collected from the East River and the Phase 2 reference areas). Patterns of TPAH (34) and TPAH (17) in surface water were very similar to one another. While Pb was detected during

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[148] The spatial pattern of dieldrin, which was only evaluated for surface sediment and tissue, is also similar to that of TPCB in surface sediment.

wet weather conditions, its spatial pattern is similar to that of Cu; that is, concentrations increase with distance from the mouth of the creek to upstream reaches. C19-C36 was not analyzed in surface water.

- 2,3,7,8-TCDD was not measured in **shallow porewater, mid-depth porewater, or groundwater**, and Pb was infrequently detected in mid-depth porewater and groundwater. C19-C36 was not measured in shallow porewater; it was measured in mid-depth porewater and groundwater, and concentrations in those two media were largely non-detect, with concentrations of detected samples showing no pattern and being within the ranges of MDLs for the non-detect samples. In contrast, TPAH (17), TPAH (34), and TPCB were frequently detected in shallow porewater, mid-depth porewater, and groundwater. Cu was also detected in these media but less frequently than TPAH (17) and TPCB (and generally more frequently than Pb). Pb was detected in shallow porewater, with a spatial pattern similar to that of Cu (see Figures 4-203 for Cu and 4-205 for Pb).
- The pattern and spatial trends for 2,3,7,8-TCDD (see Figures 4-279 through 4-286) and dieldrin (see Figures 4-303 through 4-310) in the **tissue** of all biota and tissue types sampled are broadly similar to those of TPCB (see Figures 4-271 through 4-278), with dieldrin showing more similarities to TPCB than 2,3,7,8-TCDD. Similarly, the pattern and spatial trends for Pb (see Figures 4-295 through 4-302) in all biota sampled and all tissue types are similar to Cu (see Figures 4-287 through 4-294). Like TPCB, the patterns and spatial trends of tissue concentrations for 2,3,7,8-TCDD and dieldrin in general reflect surface sediment concentrations. Likewise, the lack of a strong relationship between tissue Pb concentrations and surface sediment Pb concentrations is similar to the lack of a strong relationship between tissue Cu concentrations and surface sediment Cu concentrations. TPAH (34) and C19-C36 were not included in the analyte list for tissue samples.

TPCB and 2,3,7,8-TCDD are both bioaccumulative compounds that are important risk drivers through a bioaccumulative pathway from sediment to tissue. Although the distributions of these two contaminants in sediment and tissue are broadly similar (as discussed in the preceding text), there are some differences. As described in Section 6.6, empirical data on surface sediment and/or tissue will be evaluated in the

FS to develop an approach for evaluating the extent to which each alternative may reduce risk from dioxins/furans, in addition to TPCB.

4.12.3 Summary

The distributions of contaminants in surface sediment and other media presented in Sections 4.2 through 4.10, and the comparisons among them by class presented in Sections 4.12.1 and 4.12.2, describe the nature and extent of contamination in the Study Area. The contaminants evaluated for nature and extent exhibit broadly similar spatial patterns by contaminant class (i.e., hydrocarbons, bioaccumulative organics, and metals). Within these classes, the contaminants exhibit correlation and collocation spatially (to varying degrees and over varying spatial scales). The concentration gradients do vary among contaminants within a given chemical class in some cases. Regardless of similarity in spatial distribution, the contaminants within a class do not all contribute to ecological and human health risk to the same degree: the primary drivers of ecological and human health risk are hydrocarbons and PCBs, with Cu and other constituents also contributing to ecological and human health risk. As such, the subsequent sections of the RI focus on TPAH (17), TPCB, and Cu as representative contaminants from each of these three classes for evaluations of sources (see Section 5), fate and transport (see Section 6), and the overall CSM (see Section 8).[149]

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[149] However, going forward into the FS, additional contaminants will continue to be considered; for example, due to differences in hydrocarbon distributions, TPAH (17) cannot be considered a surrogate for other hydrocarbons, so the FS will consider the other hydrocarbon groups (i.e., TPAH [34] and C19-C36) individually.

5 SOURCES

One objective of the Newtown Creek RI stated in the RI/FS Work Plan (AECOM 2011) is to identify and quantify significant ongoing contaminant loads to the Study Area and, "to the extent of the available information, sources of such loadings." Potential source pathways to the Study Area include the following: point sources and overland flow that discharge to the surface water; groundwater that discharges to the subsurface sediment at the native material interface and laterally through vertical permeable shorelines to the surface water (i.e., lateral discharge); the East River; and other non-point sources[150] (i.e., shoreline erosion, overwater activities, atmospheric deposition, and shoreline seeps) that may introduce contaminants, including NAPL in some cases, to the Study Area surface water.[151] Based on discussions in Section 3 and as reflected in the sediment data, it is clear that the Study Area was subject to a long and dynamic history of chemical loading and deposition. This section presents an evaluation of these ongoing sources; it does not discuss or attempt to quantify historical sources (i.e., those are discussed in Section 3).

As presented in Sections 4.1.2 and 4.12.3, the discussion of sources in this RI Report focuses on concentration data for TPAH (17), TPCB, and Cu. Although this section and the associated tables, figures, and appendices identify many ongoing sources of contamination, neither this section nor this RI Report is intended to be an exhaustive list of sources of contamination. Identification and evaluation of potential sources is still ongoing.

The magnitude of a chemical source is measured as a "load," which is defined in units of mass per unit of time (e.g., kilograms per year [kg/year]). Estimates of current loads are a key determinant of the effect of sources on the Study Area, but they have to be evaluated in combination with other fate and transport processes in order to support the development of

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[150] Overland flow was also evaluated as an "other non-point source" in the draft SSAM. As discussed in the draft SSAM and in Section 5.1.2, for the purposes of this RI, loads in stormwater discharges to the Study Area, including overland flow and point source discharges, are evaluated through the Phase 2 point sources sampling program and load evaluation. The flow estimates from the 2015 geo-neutral point source model (presented in Section 5.1.2) are not differentiated between stormwater that discharges to the creek as overland flow versus stormwater that discharges as a point source. As a result, it is not necessary to make a distinction between stormwater and overland flow for load calculations.

[151] External sources to the Study Area (to both sediment and surface water) are discussed in Section 5. Internal sources, such as porewater advection and gas ebullition-facilitated transport, are considered fate and transport processes and are discussed in Section 6.

the CSM to ultimately guide evaluations of remedial alternatives in the FS. Further discussion of sources in combination with fate and transport and the CSM are presented in Sections 6 and 8. For some sources discussed in this section (point sources, groundwater discharge to the base of the Study Area,[152] and atmospheric deposition), quantitative estimates of loads are presented for TPAH (17), TPCB, and Cu. For other sources (lateral groundwater discharge,[153] shoreline erosion, overwater activities, and shoreline seeps), a qualitative discussion of the available information from which these sources were assessed is provided. Similar to the discussion of the nature and extent of contamination in Section 4, the results of source evaluations in this section are discussed for the following reaches, CM 0 – 1, CM 1 – 2, CM 2+, and each individual tributary (see discussion of Study Area reaches in Section 4-1 and Figure 4-3).

5.1 Point Sources and Overland Flow

Solids and chemical loads that enter the Study Area from point source discharges and overland flow represent ongoing sources to Newtown Creek. Estimated current loads, based on data collected from point source discharges and overland flow to Newtown Creek during the Phase 2 sampling program, are presented in this section.

As discussed in Section 3.2.8, types of current point source discharges to the Study Area are individually permitted discharges, WWTP treated effluent overflow, CSOs, and stormwater (including overland flow). The individually permitted discharge includes discharges of treated effluent from groundwater remediation and dewatering systems,[154] stormwater from industrial sites, hydrostatic test water, and discharge from secondary containment systems, each of which has an individual permit for discharge to the creek, rather than a general permit or no permit at all. Stormwater discharges (both point sources and overland flow) are

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[152] The base of the Study Area is defined as the interface between sediment and native material, as well as between sediment and fill.

[153] Shallow lateral groundwater discharge inputs to Newtown Creek have not been empirically characterized; thus, USEPA is planning a study to further characterize shallow lateral groundwater discharge along the shoreline of Newtown Creek. The stated objective of the USEPA study is to collect empirical data to achieve sufficient characterization of shallow lateral groundwater discharge to support the FS and reduce uncertainty in the current lateral groundwater discharge estimate.

[154] Treated effluent from groundwater remediation and dewatering systems (groundwater effluent) is groundwater that has been pumped from the subsurface, treated (degree of treatment varies), and discharged to Newtown Creek through an outfall. Descriptions of these discharges are provided in Section 3.2.8.

discharges of precipitation runoff from private, commercial, and industrial sites—and from roads and open spaces. More than 300 private and municipal outfalls have been documented along Newtown Creek and its tributaries, some of which may be abandoned or no longer in use (see Figures E2-4a through E2-4j). An inventory of known current point source and overland flow discharges is presented in Section 2.1 of Appendix E. Historical point source discharges and overland flow are also discussed in Section 3, and additional information about historical discharges is included in Section 4.2.4 of Appendix E. Data collected during the Phase 2 point sources sampling program are presented in Section 7 of Appendix Bi and discussed in this section. Load calculations and supporting information are included in Section 3 of Appendix E, and a summary is presented in this section.

5.1.1 Point Source Discharge Categories

During the RI, the point source discharges were divided into categories of similar types of discharges to aid in the selection of sampling locations and the estimation of loads to the Study Area. The point source categories are listed here and described in the following subsections:

- Category 1 – Individually permitted stormwater and wastewater discharges
- Category 2 – CSOs and WWTP treated effluent overflow
    - Category 2A – CSOs
    - Category 2B – WWTP treated effluent overflow
- Category 3 – Other stormwater discharges
    - Category 3A – MS4 and major stormwater discharges
    - Category 3B – Highway drains
    - Category 3C – Direct discharges from individual sites

5.1.1.1 Category 1 – Individually Permitted Stormwater and Wastewater Discharges

Eight[155] Category 1 discharges to the Study Area were identified based on USEPA's online databases; these discharge locations are shown in Figures E2-4a through E2-4j. Category 1

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[155] The discharge from Getty Terminals (DAR No. 47) regulated under SPDES Permit No. NY0028452 was permitted in 2012; however, following the sale of the property and change in site use in 2014, this discharge is no longer permitted.

discharges are individually permitted stormwater and wastewater discharges (e.g., treated groundwater effluent from remediation and dewatering systems). Individual permits are issued by NYSDEC under SPDES, New York State's USEPA-approved state regulatory program. Individually permitted sites are required to monitor their discharge for parameters identified in the site's permit and to report the monitoring data to regulatory agencies in DMRs. Individual permits are written for a specific site, and the requirements of each permit vary.

5.1.1.2 Category 2 – Combined Sewer Overflows and WWTP Treated Effluent Overflow

Category 2 discharges include overflows from the combined sewer system and treated effluent overflow from the Newtown Creek WWTP.[156] Category 2 discharges to Newtown Creek are regulated by SPDES permits for the Newtown Creek WWTP (Permit No. NY0026204) and Bowery Bay WWTP (Permit No. NY0026158) (NYSDEC 2015b, 2015c). The following subcategories were established to further classify Category 2 discharges as part of the RI:

- **Category 2A – Combined sewer overflows**. Twenty-two CSO outfalls discharge to the Study Area, as shown in Figures E2-4a through E2-4j. Portions of the Newtown Creek sewershed are served by municipal combined sewer systems. In these areas, stormwater and wastewater discharges from private and public sites enter the same pipe and are conveyed to the WWTP. When the hydraulic capacity of the combined sewer infrastructure is exceeded, combined sewage will discharge directly to the Study Area through a CSO outfall. The occurrence of a CSO at a particular outfall is a function of multiple factors, including the amount and intensity of precipitation, hydraulic conditions in regulators and the interceptor sewers, and tide stage of the creek. Bypasses may also occur during dry weather when combined sewer infrastructure malfunctions (NYCDEP 2011c, 2012b, 2013a, 2014b, 2015, 2016a). These dry weather bypasses are infrequent, short-term discharges and result in the discharge of a small volume compared to that of the wet weather CSO discharges.
- **Category 2B – WWTP treated effluent overflow**. The one Category 2B outfall is for treated effluent from the Newtown Creek WWTP. Treated WWTP effluent is normally discharged to the East River and is regulated by SPDES Permit No. NY0026204.

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[156] Historically, NYCDEP and NYSDEC used the term Water Pollution Control Plant (WPCP); however, in the 2015 permits, these facilities are referred to as Wastewater Treatment Plants (WWTPs).

However, under certain high-flow (wet weather) conditions and during high tide, this treated effluent is discharged to Whale Creek via a high-relief outfall (NCB-002; see Figures E2-4a through E2-4j) (Greeley and Hansen et al. 2010). The flow splitting between these two outfalls (to the East River and Whale Creek) is based on hydraulic conditions and does not require operator control. Tidal elevation and total plant inflows determine when the treated effluent is directed to Whale Creek.

5.1.1.3 Category 3 – Other Stormwater Discharges

Category 3 discharges are stormwater discharges that do not have an individual permit. Category 3 discharges include stormwater discharges from sites that have an MSGP and municipal discharges for which there is an NYC MS4 permit,[157] as well as stormwater discharges that are not regulated. Category 3 discharges include stormwater point source discharges and overland flow from individual sites directly to the Study Area, from municipal infrastructure to the Study Area, or a combination of both. The majority of the more than 300 outfalls that have been identified adjacent to the Study Area are stormwater outfalls.

The following subcategories were established within Category 3 for the RI point sources evaluation:

- **Category 3A – MS4 and major stormwater discharges**. There are 21 Category 3A outfalls, as shown in Figures E2-4a through E2-4j. MS4s, major stormwater outfalls, and overland flow convey runoff from multiple sites. In some cases, large portions of the Newtown Creek drainage basin discharge at a single Category 3A outfall or overland flow location. In several areas within the drainage basin, stormwater runoff from private sites and roadways drains to MS4 infrastructure and is then discharged to the Study Area. MS4s are owned by NYC and are regulated under SPDES Permit No. NY-0287890.

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[157] Individual permits are specifically developed for the needs of a single facility. General permits are written to apply to multiple discharges within a type of activity or geographic area that have similar environmental impacts. In New York State, an MSGP (No. GP-0-12-001) for stormwater discharges associated with industrial activity has been issued by NYSDEC. The MSGP is applicable to 31 categories of industrial activities, including scrap recycling and waste recycling facilities, land transportation, and warehouse facilities. Facilities that fall within one of these 31 categories may apply for coverage under the permit. On August 1, 2015, NYSDEC issued a new permit (SPDES Permit No. NY-0287890), which covers NYC-owned MS4 discharges (NYSDEC 2016b).

- **Category 3B – Highway drains**. There are 25 Category 3B outfalls, as shown in Figures E2-4a through E2-4j. Stormwater runoff from major roads and highways, including the Brooklyn Queens Expressway (I-278) and the Long Island Expressway (I-495), drains directly to the Study Area or to municipal infrastructure (either combined or separate pipes). The highways are under New York State Department of Transportation jurisdiction, and local roads are generally under the jurisdiction of New York City Department of Transportation.
- **Category 3C – Direct discharges from individual sites**. For the purposes of this RI, discharges that are not documented as Category 1, 2A, 2B, 3A, or 3B are considered to be Category 3C discharges draining individual sites. There are more than 200 Category 3C outfalls documented in the outfall inventory (see Figures E2-4a through E2-4j). This subcategory includes stormwater from individual properties that is discharged directly to the Study Area as point sources or as overland flow. Examples of Category 3C discharges are stormwater from facilities (e.g., warehouses, transfer stations, ready-mix concrete manufacturers) and parking lots near the creek that drain directly to the creek, through privately owned stormwater infrastructure. Generally, Category 3C discharges are not regulated, but in some cases, the discharge is regulated by the MSGP.

5.1.2 Flow Data

NYCDEP would not allow the NCG to install flow meters in municipal infrastructure during the Phase 2 point sources sampling program. As a result, only concentration data were collected as part of the Phase 2 point sources sampling program (see the discussion in Section 5.1.3). Because flow was not measured at the time samples were collected, flows were estimated using a geo-neutral version of the NYCDEP InfoWorks model, which was calibrated using data from flow meters installed in the New York City sewer system by NYCDEP and used to predict flows from CSOs and MS4s,[158] or were based on information provided by property owners or from data extracted from published reports as described in this section.

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[158] Flow meters installed by NYCDEP in five CSO outfalls (BB-026, NCB-015, NCB-083, NCQ-029, and NCQ-077) between March 2014 and March 2015, as part of a City-wide flow monitoring program, were used by NYCDEP in the development of the revised 2015 version of the geo-neutral point source model.

CSOs and stormwater. The primary source of flow information used for the point sources evaluation was the estimates generated by the landside modeling effort performed by NYCDEP (NYCDEP point source model) in support of the city-wide LTCP for CSOs (NYCDEP 2007a, 2007b, 2007c, 2007d, 2011b; AECOM 2017).[159] As discussed in Section 3.3 of Appendix G, the 2015 geo-neutral version of the NYCDEP point source model (referred to hereafter as the 2015 geo-neutral point source model) was used to predict fresh water inflows to Newtown Creek for use in the hydrodynamic, sediment transport, and CFT models.[160] The 2015 geo-neutral point source model was used to estimate the arithmetic average annual CSO and stormwater flow (including overland flows[161]) for 2008 through 2012,[162] for the purposes of calculating the load estimates presented in this section. Flows for stormwater discharges that were not identified individually in the model were estimated based on relative drainage area. The arithmetic average annual CSO discharge to the creek is 1,600 MG. Arithmetic average annual stormwater discharges are 830 MG, 330 MG of which is discharged from MS4s.

Annual discharge of treated effluent from groundwater remediation and dewatering systems was estimated using data reported by individually permitted sites in permit-required DMRs and associated documentation (e.g., SPDES permit applications). These discharges were not included in the 2015 geo-neutral point source model, but have been added to the hydrodynamic model (see Section 4.4.4 of Appendix G).

WWTP treated effluent overflow. NYCDEP provided estimates of the number of total discharges and average annual flows from NCB-002 for 2008 to 2012 on April 10, 2014

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[159] The 2017 LTCP (AECOM 2017) was not referenced specifically for flow rate information used in the point sources evaluation because RI sampling was conducted in 2014 and 2015, prior to the release of the LTCP report.

[160] The 2015 geo-neutral point source model (provided by NYCDEP in 2016) was used to estimate flows for the RI load estimates. As discussed in Section 2.1.3 of Appendix E, prior to sampling, the 2013 geo-neutral point source model estimates were used in the evaluation of point source and overland flow discharges and the selection of sampling locations. The 2015 geo-neutral point source model may be revised or augmented in the future to improve predictions or estimates of fresh surface water inflow to the Study Area from various discrete subbasins (e.g., direct drainage areas).

[161] The 2015 geo-neutral point source model does not differentiate between stormwater that discharges to the creek as overland flow and stormwater that discharges as a point source. As a result, it is not necessary to make a distinction between stormwater and overland flow for load calculations.

[162] This 5-year period is representative of a typical range of precipitation conditions, as discussed in Section 4.1.1 of Appendix E. In Appendix E, flows were also evaluated over a 20-year period. Comparisons between flows for this 5-year period (2008 through 2012) and the 20-year period are discussed in Section 4.1.1 of Appendix E.

(see Section 3.8 of Appendix G).[163] These estimates indicate that treated effluent from the Newtown Creek WWTP discharges to Whale Creek via NCB-002 (Station ID NCB002; see Figure 5-1) from 38 to 50 times per year, with a total estimated annual discharge to Whale Creek ranging between 500 and 1,030 MG and an arithmetic average annual discharge of 730 MG.

ExxonMobil Greenpoint Remediation Project (DAR No. 53; two discharges). Groundwater treatment system effluent is discharged at NY0267724-001 (Station ID EM001A) and NY0267724-002 (Station ID EM002), as presented in Figure 5-1. Monthly total discharge volumes supplied in January 2008 through December 2012 DMRs were used to estimate annual treated groundwater effluent discharge volumes for each outfall. January 2008 to December 2012 was chosen to be consistent with the time frame used for stormwater and CSO flow estimates. Arithmetic average annual discharges from NY0267724-001 and NY0267724-002 were reported as 130 MG and 160 MG, respectively. Stormwater discharged from NY0267724-001 (Station ID EM001B) is not added to the flows used in the RI modeling efforts, because it is already accounted for as part of the stormwater discharge simulated by the 2015 geo-neutral point source model.

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[163] In response to a request from the NCG, the annual discharge volumes for the WWTP treated effluent overflow to Newtown Creek were estimated by NYCDEP, and the results were initially transmitted to the NCG and USEPA on April 10, 2014. As discussed in Section 3.8 of Appendix G, NYCDEP conducted a conference call with USEPA and the NCG on April 1, 2015, to discuss the methodology used to calculate the average annual WWTP treated effluent overflow discharge volumes and transmitted the average annual discharge volumes for the WWTP treated effluent overflow for 2008 through 2014 to USEPA and the NCG again on April 1, 2015.

Con Edison – 11th Street Conduit (DAR No. 110). Groundwater that infiltrates into this utility conduit, plus a small quantity of stormwater that enters the riser shaft, is passed through an oil-water separator and discharged to the creek at NCB-0201138 (Station ID CE11SC; see Figure 5-1). Measured flows were not provided on DMRs for this site. Based on information provided by facility personnel and field observations of the outfall, the annual discharge is estimated to be approximately 6.2 MG.[164]

Estimates of the relative annual volume of discharge to Newtown Creek from CSOs, WWTP treated effluent overflow, stormwater (including overland flow), and treated effluent from groundwater remediation and dewatering systems are shown in Figure 5-2. Based on the sources of information and methods of estimation described in this section, the annual point source (and overland flow) discharge to the Study Area is approximately 3,400 MG. CSOs contribute approximately 46% of the total point source (and overland flow) discharge to the Study Area, and the WWTP treated effluent overflow contributes approximately 21%. Stormwater (including overland flow) accounts for 24%, approximately 9.5% of which is associated with MS4 discharges.[165] The remaining approximately 8.4% of the overall point sources (and overland flow) discharge to the Study Area is treated groundwater effluent from remediation and dewatering systems (i.e., ExxonMobil Greenpoint Remediation Project [DAR No. 53], Con Edison – 11th Street Conduit [DAR No. 110], and Buckeye Pipeline Facility [DAR No. 106]), which is discharged from individually permitted outfalls.[166,167]

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[164] An average pumping rate of 85,000 gallons per day was reported in the facility's SPDES permit application. As discussed in Section 2.1.3.3 of Appendix E, this value is based on the pump's design flow rate (230 gallons per minute) and the approximate frequency at which the pump operates, for which a conservative estimate of 15 to 20 minutes each hour was provided by site personnel. However, the pump is activated by a level float, and the actual duration of pumping and volume of discharge are not measured and likely vary temporally. For the purposes of the loadings presented in this RI Report and based on field observations of the outfall and insights from preliminary CFT modeling, it is estimated that the flow rate is likely lower than the permit value by at least a factor of five (see Section 4.1.3.2 of Appendix E for more detail). As such, the annual average flow rate used for loading estimates for this discharge was estimated to be 17,000 gallons per day (and it should be recognized that there is a high degree of uncertainty associated with this estimate).

[165] This estimate is based on the arithmetic average of the annual flow for 2008 to 2012. Annual stormwater (including overland flow) and CSO discharges were estimated by Anchor QEA using the 2015 geo-neutral point source model (see Section 3 of Appendix G). Annual flows for the WWTP treated effluent overflow were provided by NYCDEP.

[166] This estimate is based on flow data provided in DMRs for individual facilities, as was discussed in Section 5.1.2.

[167] Remedial pumping at the Buckeye Pipeline Facility (DAR No. 106) operated from 1987 to October 2012. In March 2015, NYSDEC discontinued the SPDES permit, due to the removal of the treatment system (GES 2017).

5.1.3 Phase 2 Point Sources Sampling Program

To estimate solids and chemical loads from a point source discharge, the following two types of data are needed:

- Flow rate (i.e., the volume of water discharged per unit time)
- Solids or chemical concentrations (i.e., mass per unit volume [M/L^3]) of the discharge

The purpose of the RI point sources sampling and analysis effort was to collect these data, as discussed in this subsection. Concentration data used to estimate point source loads were obtained by sampling point source discharges and overland flow. Sampling was conducted to meet the objectives of the field program outlined in Section 5 of the Phase 2 RI Work Plan Volume 2 (Anchor QEA 2014b), using the methods described in Section 10 of the Phase 2 FSAP Volume 2 (Anchor QEA 2014d).

5.1.3.1 Objectives

The first objective of the point sources sampling and analysis effort was to provide data that can be used to evaluate the composition (i.e., loads of various chemicals and solids) of potentially significant point source discharges and overland flow to the Study Area. The second objective was to develop an improved understanding of the relative magnitude and spatial and temporal variation in point source discharges and overland flow to the Study Area, to refine elements of the CSM and support future evaluation of remedial alternatives in the FS. As documented in the Phase 2 RI Work Plan Volume 2, flows and concentrations of stormwater discharges and CSOs are expected to vary widely due to several factors, including (but not limited to) characteristics of storms and associated runoff (e.g., antecedent dry periods; storm precipitation amounts, intensity, and durations; collection system and interceptor system characteristics; and presence, condition, and proper functioning of source controls) (Anchor QEA 2014b). To fully characterize this variability, a large number of water samples taken over the course of many years and many types of storms, pollutant sources, and runoff conditions would be necessary. However, such an approach was not necessary to meet the objectives and schedule for the RI/FS, because, although variability is recognized, the sampling program was designed to understand ranges and average concentrations.

Consistent with the first objective of the Phase 2 point sources sampling program, the primary focus of the point sources sampling effort was to collect whole-water samples and measure the total solids and chemical concentrations discharging to the creek during a sampling event. To mitigate issues related to temporal variability during the sampling events (short-term variability), composited whole-water samples were collected over the duration of the sampling event to provide data needed to estimate event mean concentrations (EMCs). To provide a consistent compositing method across the sampling locations, time-based compositing was conducted. Time-based compositing provides an average of the contaminant concentrations collected at several snapshots in time during a single storm sampling event. This approach (i.e., whole-water sampling with time-based compositing) is similar to a sampling approach used at the Gowanus Canal Superfund Site by NYCDEP (LBG 2012, 2013) and the National Grid Site (Woodard and Curran 2012). Longer-term variability (over the course of a year) was addressed by sampling multiple discharge events and multiple locations.

To estimate point source loads, EMCs were applied in a fixed concentration load methodology, as further discussed in Section 5.1.4. The point sources sampling and load analyses approach was a practical alternative to a time-varying methodology, such as a buildup/wash-off methodology,[168] which provides a greater level of temporal detail, but is more resource intensive. A fixed concentration methodology was appropriate for RI/FS purposes to achieve the objectives of the point sources evaluation.

The locations of discharges sampled during the Phase 2 sampling program are shown in Figure 5-1. Results of sampling are presented in Section 7 of Appendix Bi and Attachment E-A of Appendix E. Rationale for the selection of sampling locations and measurement methods used during the point sources sampling effort is discussed in Section 2.2 of Appendix E.

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[168] Buildup/wash-off models were developed following studies conducted in the 1960s and 1970s that established that solids (dust and dirt) accumulate on surfaces in urban watersheds during dry weather periods between storms. Buildup/wash-off models include functions to estimate both the accumulation (buildup) rate of solids and the rate at which the accumulated solids (and associated pollutants) wash off (i.e., erode or dissolve into runoff). Buildup and wash-off rates have been shown to vary depending on numerous factors, including land use, street cleaning methods, and climate. Site-specific flow and concentration data collected throughout hydrographs for a variety of storms are needed to support buildup/wash-off models (Rossman and Huber 2016).

5.1.3.2 Locations and Types of Sampling

Point source samples were collected from 31 point source discharges and from the influent at the Newtown Creek WWTP during 8 dry weather and 15 wet weather sampling events, between June 2014 and December 2015. Table 5-1 presents the number of samples collected at each location.

Twenty samples were collected from seven CSO sampling locations: NCB083, NCQ077, NCB015, BB026, NCQ029, BB009, and NCB022. Three samples were collected from each location, with the exception of NCB022 (sampled one time) and BB026 (sampled four times). Discharges from these seven CSOs account for approximately 97% of the total CSO discharge volume to the Study Area. Three samples were also collected from the overflow at the Newtown Creek WWTP (i.e., NCB002), which discharges WWTP treated effluent.

Three stormwater samples were collected from each of the following locations, with the exception of Queens District 5/5a Garage (DAR No. 45) (QDG002; two samples collected) and NCQ632 and NCQ633 (one sample collected at each location), for a total of 58 samples:

- **Four individually permitted stormwater discharges (Category 1)**. EM001B[169] (ExxonMobil Greenpoint Remediation Project [DAR No. 53]), MBT001 (Motiva Brooklyn Terminal [DAR No. 50]), BPBT001 (BP Products N America Brooklyn Terminal [DAR No. 48]),[170] and QGD002 (Queens District 5/5a Garage [DAR No. 45]).

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[169] Discharge from NY0267724-001 was sampled at two stations, EM001A and EM001B. During reconnaissance, sampling locations were identified that allowed stormwater and groundwater effluent to be sampled separately. Groundwater treatment effluent flows were sampled at EM001A, and stormwater flows were sampled at EM001B.

[170] BP Products N America Brooklyn Terminal was sold to Kinder Morgan Operating L.P. "D", a Delaware limited partnership effective February 2, 2016. Nonetheless, to maintain consistency with the draft DAR, this facility is referred to as "BP Products N America Brooklyn Terminal."

- **Six MS4 discharges (Category 3A)**. NCB631, NCQ632B, NCQ637, NCB629, NCQ632,[171] and NCQ633.[172]
- **Three stormwater discharges of runoff from multiple sites (Category 3A)**. BB610 (Amtrak Sunnyside Yard [DAR No. 102] and Metropolitan Transportation Authority LIRR rail yard), O185 (ESF Trucking, Mendon Trucking, and Cadogan Tate), and MA001 (Meeker Avenue overland flow).
- **One discharge of runoff from a highway (Category 3B)**. LIE001 (Long Island Expressway).
- **Seven stormwater discharges from individual private sites (Category 3C)**. HN001 (Hugo Neu [DAR No. 125]), MCL001 (Maspeth Concrete Loading Corp. [DAR No. 210]), RAD001 (RAD I [DAR No. 41]), FLH001 (Former Laurel Hill Site [DAR No. 16]), WM001 (Waste Management of NY/Steel Equities [formerly POW] [DAR No. 56]), NG001 (Greenpoint Energy Center [DAR No. 32]), and MP001 (Malu Properties/Former Ditmas Oil/Former Gulf Oil [DAR No. 123]).

In addition to CSO, WWTP treated effluent overflow, and stormwater (including overland flow) samples, three samples were collected from each of three individually permitted groundwater effluent locations, for a total of nine samples. Treated groundwater effluent discharge from the ExxonMobil Greenpoint Remediation Project (DAR No. 53) system was sampled in two locations

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[171] Based on information provided by NYCDEP, traffic control was initially believed to be infeasible at the NCQ-632B manhole, which was located in the roadway. Once traffic control was determined to be feasible, USEPA requested that the manhole downstream of the junction of Grand and Page Place (NCQ-632B) replace the original sampling manhole (NCQ-632). According to NYCDEP drainage maps, sampling at the NCQ-632B location captures approximately 64 acres of the 80-acre NCQ-632 drainage basin, including the runoff from Page Place and a large portion of Grand Avenue. Sampling location NCQ-632 was sampled once before being removed from the sampling program.

[172] During planning for Phase 2, the drainage basin for Outfall NCB-631 was identified as a potential point sources sampling location, but was ultimately not selected for sampling because, according to information provided by NYCDEP, only a small portion of the drainage basin would be captured due to tidal inundation in the drainage infrastructure. On December 19, 2014, NYCDEP provided Anchor QEA and USEPA with modified drainage basin information for NCB-631, showing a larger 125-acre basin with a non-tidally inundated potential sampling manhole, conveying the runoff from approximately 50 acres of the drainage basin. Based on this new information and the knowledge that traffic control permits could be obtained, USEPA decided the drainage basin for Outfall NCB-631 should be targeted for point sources sampling. Because the NCQ-633 sampling location captured only approximately 6 acres of a 37-acre drainage basin (according to NYCDEP), sampling location NCB-631 replaced NCQ-633 in the Phase 2 point sources sampling program. Sampling location NCQ-633 was sampled once before being removed from the sampling program.

(EM001A and EM002), representing two separate permitted treated groundwater effluent discharges to the creek. Samples were also collected from the groundwater effluent discharge for Con Edison – 11th Street Conduit (DAR No. 110) (CE11SC).

Samples were collected from Newtown Creek WWTP influent flows during wet and dry weather conditions. The WWTP influent does not discharge to Newtown Creek, and these data were not used directly to estimate loads. WWTP influent data were used to evaluate variability in CSO discharges and are discussed as part of the uncertainty analyses presented in Section 4 of Appendix E.

The Phase 2 point sources sampling program included collection of whole-water samples for analysis of all target compounds, as well as field-filtered dissolved phase samples for metals, at each location. Additionally, 15 particulate and dissolved phase samples (i.e., samples in which organics present in the dissolved and particulate phases were analyzed separately) were also collected at a subset of point source locations, as well as the influent at the Newtown Creek WWTP.[173]

5.1.3.3 Sampling Results

The whole-water (i.e., total) and particulate phase[174] concentrations in samples collected during the Phase 2 point sources sampling program are presented in Attachment E-A of Appendix E for all analytes. Whole-water and particulate phase concentrations for TPAH (17), TPCB, and Cu, along with concentrations of key water quality parameters (TSS, f_{OC}, and DOC), measured in point source samples are presented in Figures 5-3 through 5-11 and in Tables 5-2 through 5-10. The results are discussed in the following subsections. WWTP influent results are discussed in Section 4.2.2.2 of Appendix E.

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[173] Dissolved/particulate phase samples were collected from the following locations: QDG002, BB026, NCB015, NCB032, NCQ077, NCB002, NCB629, MA001, BB610, NCQ637, LIE001, HN002, and MCL002, and two samples were collected at the WWTP influent. Additional discussion of dissolved/particulate phase sampling is provided in Appendices Bi and E.

[174] Chemicals within whole-water samples exist primarily in particulate and dissolved phases, where the former corresponds to chemicals that are sorbed to, or associated with, particulate matter (i.e., suspended sediment) within the samples.

Except where otherwise noted, the discussions of particulate phase concentrations for TPCB and TPAH (17) in this section are based on the combination of measured and estimated values. For the 15 samples in which organic compounds were measured in particulate and dissolved phases, the particulate phase data are reported directly, and total concentrations for those samples are calculated by summing the individual dissolved and particulate phase concentrations. Particulate phase metals concentrations (reported on a dry weight basis) were calculated by subtracting the dissolved concentration from the whole-water concentration and dividing by the TSS concentration. For samples in which only the total concentrations of organic compounds were measured, particulate phase concentrations were estimated using well-established methods for evaluating partitioning. This calculation included using site-specific partition coefficients (discussed in Section 6.4.1.3 of this report and Attachment E-C of Appendix E) and the TSS concentration of each sample. Results of this calculation were checked against the subset of locations at which particulate fractions were measured directly. A detailed discussion of the calculations used to generate estimates of particulate phase concentrations for whole-water samples is provided in Attachment E-C of Appendix E.

Consistent with the discussion in Section 4.12, the point sources sampling results are presented in the following subsections for key water quality parameters (TSS, f_{OC}, and DOC), followed by the chemicals being used for quantitative source evaluations in the RI Report: TPAH (17), TPCB, and Cu. As noted previously, point sources sampling results for all chemical analytes are provided in Attachment E-A of Appendix E.[175]

5.1.3.3.1 TSS

TSS concentrations are presented in Figure 5-3 and summarized in Table 5-2. Concentrations differ by discharge type, as follows:

- **CSO** TSS concentrations have an arithmetic average of 96 mg/L, with a range of 27 to 220 mg/L).

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[175] Specifically, figures showing concentrations by point source location and tables of concentration summary statistics by discharge type are provided for the full list of chemical analytes in Attachment E-A of Appendix E. These analytes include TPAH (34), 2,3,7,8-TCDD, and Pb. Point sources samples were not analyzed for C19-C36.

- **WWTP treated effluent overflow** TSS concentrations are lower than concentrations measured in CSO discharges, with an arithmetic average of 21 mg/L and a range of 13 to 35 mg/L.
- **Stormwater (including overland flow)** TSS concentrations (including Category 1 stormwater samples and Category 3 stormwater samples) have an arithmetic average of 120 mg/L, with a range of non-detect (5.0 mg/L) to 760 mg/L. With the exception of QDG002, TSS concentrations in samples collected from the Category 1 individually permitted stormwater locations are lower than concentrations in samples collected from Category 3 locations. As with the stormwater samples, the arithmetic average TSS concentration for MS4 samples is also 120 mg/L; however, the range of TSS concentrations in MS4s is 15 to 310 mg/L. The highest TSS concentrations occur in samples collected from a Category 3A overland flow location (MA001; 760 mg/L) and a Category 3C point source location (MCL001; 590 mg/L).
- **Treated groundwater effluent** TSS concentrations are generally the lowest among all point source samples, with an arithmetic average of 8.4 mg/L and a range of non-detect (5.0 mg/L) to 19 mg/L.

5.1.3.3.2 Organic Carbon

OC in point source and overland flow samples is evaluated based on POC and DOC data, as summarized in the discussion that follows.

POC in water samples was evaluated by normalizing it to the TSS concentration, which provides the f_{OC} of the suspended solids. Units are wt% on a dry weight basis, consistent with evaluations of surface water and measurement of sediment. Arithmetic average f_{OC} values in point sources and overland flow samples are shown in Table 5-3 and Figure 5-4. Concentrations differ by discharge type as follows:

- **CSOs** had a range of f_{OC} from 5.7 wt% to 46 wt% and an arithmetic average of 16 wt%.
- **WWTP treated effluent overflow** f_{OC} (arithmetic average = 41 wt%) was more than twice that of the CSOs and did not vary as much across sampling events, with a range of 36 wt% to 52 wt%.
- **Stormwater (including overland flow)** f_{OC} was similar to CSOs, had an arithmetic average of 16 wt%, and ranged from 1.5 wt% to 39 wt%.

- **Treated groundwater effluent** average f_{OC} was also similar to CSOs and stormwater and had an arithmetic average of 16 wt%, but samples from individual events were generally similar, or lower, and ranged from 6.6 wt% to 20 wt%.

DOC concentrations in samples are presented in Figure 5-5 and Table 5-4. Concentrations differ by discharge type, as follows:

- **CSOs and WWTP treated effluent overflow** had similar DOC concentrations, with arithmetic averages of 9.2 and 7.5 mg/L, respectively. DOC in CSOs ranged from non-detect (1.2 mg/L) to 26 mg/L, and DOC in WWTP treated effluent overflow had a smaller range of non-detect (3.0 mg/L) to 10 mg/L.
- **Stormwater (including overland flow)** DOC concentrations were highest for point source samples, with an arithmetic average of 19 mg/L and range of non-detect (0.59 mg/L) to 140 mg/L. Category 1 and Category 3 samples differed by less than a factor of 2, with arithmetic averages of 12 and 20 mg/L, respectively. The arithmetic average DOC concentration in MS4s was 16 mg/L, and the range for MS4s was non-detect (0.59 mg/L) to 62 mg/L.
- **Treated groundwater effluent** DOC concentrations are generally the lowest among all categories, ranging from 1.3 to 6.4 mg/L, with an arithmetic average of 3.8 mg/L.

5.1.3.3.3 TPAH (17)

Whole-water TPAH (17) concentrations are presented in Figure 5-6 and summarized in Table 5-5. Particulate phase concentrations are provided in Figure 5-7 and summarized in Table 5-6. Concentrations differ by discharge type, as follows:

- **CSO** TPAH (17) whole-water concentrations have an arithmetic average of 3.3 µg/L, a median of 2.5 µg/L, and a range of 0.77 to 9.6 µg/L. Particulate phase TPAH (17) concentrations in CSO samples (arithmetic average of 31 mg/kg and median of 27 mg/kg, with a range of 9.0 to 90 mg/kg) are similar to concentrations measured in samples collected from other discharge types.
- **WWTP treated effluent overflow** TPAH (17) concentrations are generally an order of magnitude lower than samples collected from CSO locations. Whole-water concentrations range from 0.25 to 0.48 µg/L, with an arithmetic average of 0.34 µg/L,

and median of 0.28 mg/kg. Particulate phase concentrations range from 4.8 to 7.7 mg/kg, with an arithmetic average of 5.9 mg/kg, and median of 5.2 mg/kg.

- **Stormwater (including overland flow)** TPAH (17) whole-water concentrations have an arithmetic average of 5.1 µg/L, a median of 2.3 µg/L, and a range of 0.041 to 66 µg/L. The highest stormwater TPAH (17) concentration occurs in a sample collected from a Category 3A major stormwater location, MA001, and is approximately 2 times greater than the next highest TPAH (17) concentration. Particulate TPAH (17) concentrations have an arithmetic average of 40 mg/kg, a median of 35 mg/kg, and a range of 2.1 to 140 mg/kg. Category 1 individually permitted stormwater locations, MS4 locations, and other Category 3 stormwater locations have similar whole-water and particulate phase TPAH (17) concentrations.
- **Treated groundwater effluent** TPAH (17) whole-water concentrations have an arithmetic average of 240 µg/L, a median of 0.20 µg/L, and a range of 0.0055 to 1,600 µg/L. Particulate phase TPAH (17) concentrations have an arithmetic average of 12,000 mg/kg, a median of 4,600 mg/kg, and a range of 51 to 37,000 mg/kg. However, the highest TPAH (17) concentrations occurred in samples collected from the Con Edison – 11th Street Conduit (DAR No. 110) dewatering system effluent (CE11SC), located adjacent to the Pulaski Bridge in Brooklyn (CM 0.567). Whole-water TPAH (17) concentrations in samples collected from CE11SC are approximately an order of magnitude higher than the next highest TPAH (17) concentrations measured in all point source samples and three orders of magnitude higher than the other two treated groundwater effluent samples (EM001A and EM002). Estimated particulate concentrations in samples collected from CE11SC are two orders of magnitude higher than concentrations measures in samples collected from the other two treated groundwater effluent locations (EM001A and EM002).

5.1.3.3.4 TPCB

Whole-water TPCB concentrations are provided in Figure 5-8 and summarized in Table 5-7. Particulate phase concentrations are shown in Figure 5-9 and summarized in Table 5-8. Concentrations differ by discharge type, as follows:

- **CSO** TPCB whole-water concentrations have an arithmetic average of 45 ng/L, a median of 27 ng/L, and a range of 4.3 to 190 ng/L. Particulate phase concentrations have an arithmetic average of 0.38 mg/kg, a median of 0.24 mg/kg, and a range of 0.099 to 1.1 mg/kg.

- **WWTP treated effluent overflow** TPCB whole-water concentrations are within the range of CSO concentrations (8.4 to 32 ng/L), with an arithmetic average of 18 ng/L (and median of 14 ng/L). The WWTP treated effluent overflow particulate phase TPCB concentration ranged from 0.38 to 1.7 mg/kg, with an arithmetic average of 0.83 mg/kg (median of 0.39 mg/kg).
- **Stormwater (including overland flow)** Category 1 locations have much lower TPCB whole-water concentrations (arithmetic average of 7.2 ng/L, median of 3.4 ng/L, and range of 0.38 to 38 ng/L) than MS4 and other Category 3 stormwater locations (arithmetic averages of 140 and 160 ng/L, respectively, medians of 50 and 28 ng/L, respectively, and combined range of 2.3 to 1,200 ng/L). Particulate phase TPCB concentrations in stormwater have an arithmetic average of 1.4 mg/kg, a median of 0.58, and a range of 0.018 to 22 mg/kg. Samples collected at location HN002 contain higher total and particulate phase TPCB concentrations than any other point source samples.
- **Treated groundwater effluent** TPCB whole-water concentrations are among the lowest concentrations measured and have an arithmetic average of 0.74 ng/L, a median of 0.41, and a range of non-detect (less than 0.14 ng/L) to 3.3 ng/L. Likewise, estimated TPCB particulate phase concentrations in treated groundwater effluent samples are among the lowest concentrations and have an arithmetic average of 0.059 mg/kg, a median of 0.039 mg/kg, and a range of 0.011 to 0.15 mg/kg.

5.1.3.3.5 Cu

Whole-water Cu concentrations are presented in Figure 5-10 and summarized in Table 5-9. Particulate phase Cu concentrations are available in Figure 5-11 and summarized in Table 5-10. Concentrations differ by discharge type, as follows:

- **CSO** Cu whole-water concentrations have an arithmetic average of 38 µg/L, a median of 28 µg/L, and a range from 6.9 to 120 µg/L. Particulate phase CSO Cu concentrations have an arithmetic average of 270 mg/kg, a median of 280 mg/kg, and range from 8.7 to 530 mg/kg.
- **WWTP treated effluent overflow** whole-water and particulate Cu concentrations are lower than CSO concentrations. Whole-water concentrations range from 10 to

16 µg/L, with an arithmetic average of 12 µg/L, and median of 10 µg/L. Particulate phase concentrations range from 120 to 250 mg/kg, with an arithmetic average and median of 190 mg/kg.

- **Stormwater (including overland flow)** Cu whole-water concentrations are lower at Category 1 permitted stormwater locations (arithmetic average of 16 µg/L, median of 5.7 µg/L, and range of 0.80 to 64 µg/L) than at non-MS4 Category 3 stormwater locations (arithmetic average of 55 µg/L, median of 41 µg/L, and range of 4.9 to 240 µg/L). MS4s have higher average Cu whole-water concentrations than other Category 3 stormwater locations, with an arithmetic average of 72 µg/L, a median of 48 µg/L, and a range of 7.5 to 240 µg/L. The highest whole-water Cu concentrations occur in samples collected at two Category 3A locations, MA001 and NCQ637. Particulate phase Cu concentrations in Category 1, MS4, and other Category 3 stormwater locations are similar to each other (arithmetic averages of 420, 370, and 550 mg/kg, respectively). The highest particulate phase Cu concentrations occur in samples collected at two Category 3C locations, RAD001 and NG001.
- **Treated groundwater effluent** Cu concentrations are among the lowest concentrations measured in point sources samples. Whole-water Cu concentrations have an arithmetic average of 2.0 µg/L, a median of 1.8 µg/L, and a range of non-detect (less than 0.30 µg/L) to 4.0 µg/L. Calculated particulate phase Cu concentrations have an arithmetic average and median of 29 mg/kg, and a range of 24 to 35 mg/kg.

5.1.3.4 Results Summary

Point source categories were developed based on available information to aid in characterizing potentially ongoing sources and the selection of sampling locations. Data collected during the Phase 2 point sources sampling program allow for the development of the CSM and are used in load evaluations presented in Section 5.1.4.

When comparing concentrations, it is important to note that a point source or overland flow discharge, regardless of the magnitude of the discharge volume, may represent a potentially significant load if it contains sufficiently elevated concentrations. However, as discussed in Section 5.1.2 and shown in Figure 5-2, annual discharge volumes among individual point source and overland flow discharges vary by more than an order of magnitude. More than half of the total point source and overland flow discharge to the Study Area occurs at three CSOs (NCQ077, NCB083, NCB015) and the WWTP treated effluent overflow (NCB002). A small difference in the concentration of a higher volume discharge, like the WWTP treated effluent overflow (NCB002; 730 MGY), results in a significant increase or decrease in the loadings to the Study Area. On the other hand, a relatively high concentration in a lower volume discharge, like the Con Edison – 11th Street Conduit (DAR No. 110) (CE11SC; 6.2 MGY), must be elevated by more than two orders of magnitude to result in a comparable increase in loading. The results of the load estimation are presented in Section 5.1.4. As presented in Figures 5-3 through 5-11 and in Tables 5-2 through 5-10, a summary of the TSS, f_{OC}, DOC, TPAH (17), TPCB, and Cu concentration data by discharge type is as follows:

- **CSOs**. Twenty CSO samples were collected from seven locations during the Phase 2 point sources sampling program. Concentrations among CSO samples are generally within one order of magnitude of each other on a whole-water and particulate phase basis, with the exception of whole-water TPCB concentrations, which span a little less than two orders of magnitude. In general, whole-water and particulate phase concentrations in CSO samples are within the mid-range of concentrations measured in samples collected from other point source discharge categories.
- **WWTP treated effluent**. WWTP treated effluent samples were collected from one location during three separate events, and one sample was collected during each event, for a total of three samples. Variability among concentrations measured in samples collected during different events was low (i.e., less than an order of magnitude). In contrast to CSOs, which are not treated, WWTP treated effluent overflow receives some treatment prior to discharge. TSS and whole-water and particulate phase TPAH (17) concentrations in WWTP treated effluent samples are generally lower than concentrations in CSO samples. However, DOC, whole-water and particulate phase Cu, and whole-water TPCB concentrations in WWTP treated effluent samples are within the range of concentrations measured in CSO samples. Particulate phase WWTP treated effluent PCB concentrations are on the higher end of the range of CSO concentrations, and f_{OC} values in WWTP treated effluent samples are among the highest measured in all samples.
- **Stormwater (including overland flow)**. In total, 58 stormwater samples were collected from 21 locations during the Phase 2 sampling program. Concentrations in stormwater

samples collected during different sampling events at the same location are generally consistent (within one order of magnitude), with the exception of f_{OC} at a few individual sampling locations (e.g., O185 and NCB629). Whole-water and particulate phase concentrations in stormwater samples are more variable than concentrations in CSO and WWTP treated effluent overflow samples, spanning two or more orders of magnitude. As expected, whole-water and particulate phase concentrations in treated stormwater discharges (BPBT001, MBT001, and EM001B) were on the lower end of the range of stormwater concentrations. Concentrations in the three samples collected from Long Island City Expressway runoff (LIE001; Category 3B) are generally in the mid-range of stormwater concentrations measured. Concentrations in samples collected from MS4 and major stormwater discharges (Category 3A) and discrete discharge from individual sites (Category 3C) vary, with some relatively high concentrations measured at both Category 3A and Category 3C locations (e.g., TPCB at HN002 and whole-water Cu concentrations at MA001 and NCQ637).

- **Treated groundwater effluent**. Three treated effluent samples from groundwater remediation or dewatering systems were collected from each of the three sampling locations during separate dry weather sampling events, for a total of nine samples. Concentrations in treated groundwater effluent samples collected at a given location were within an order of magnitude of each other. Concentrations in samples collected from treated groundwater effluent discharges are generally lower than concentrations in samples collected from other types of discharges (e.g., CSOs and stormwater), with the exception of whole-water and particulate phase TPAH (17) concentrations, which were within the mid-range of samples collected from all categories. Concentrations of whole-water and estimated particulate phase TPAH (17) in samples collected from the Con Edison – 11th Street Conduit (DAR No. 110) (CE11SC) discharge are higher than concentrations measured in all other samples.

5.1.4 Estimated Point Source Loads

Methods for calculating estimated loads from point source and overland flow discharges to the Study Area are summarized in this section. The general approach used to calculate the loads is described in Sections 5.1.4.1 and 5.1.4.2, followed by discussion of specific analyses and results for each type of discharge (e.g., CSOs, WWTP treated effluent overflow, stormwater [including overland flow], and treated effluent from groundwater remediation and dewatering systems) in Section 5.1.4.3. Additional information and other LOEs used to confirm load estimates are provided in Section 3.3 of Appendix E.

5.1.4.1 Generalized Load Equation

Estimated contaminant loads were calculated for solids and individual contaminants using Equation 5-1:

$$W = Q \times C \tag{5-1}$$

where:

W = chemical or solids load (mass per unit time [M/T])
Q = discharge rate (volume per unit time [L^3/T])
C = total chemical or solids concentration (M/L^3)

The discharge rate (Q) was obtained from the sources of information discussed in Section 5.1.2. Concentrations (C) were based on the results of the Phase 2 point sources sampling program (see Section 5.1.3.2); for the subset of samples that were analyzed separately for particulate and dissolved phases, the results from those two phases were mathematically combined to obtain the whole-water concentration.

The development of representative concentrations used to calculate loads for each discharge or type of discharge is described in subsection 5.1.4.2.

5.1.4.2 Load Estimation Approach

An objective of the Phase 2 sampling effort was to provide data to develop an improved understanding of spatial and temporal variations in loads from point source discharges. For an RI/FS-level analysis, it was not necessary to have direct measurements from every point source discharge to the Study Area. However, in order to estimate loads for all discharges, it was necessary to collect a dataset that is representative of the overall discharge to the Study Area, but also reflects unique discharges that may be potentially significant sources. The approach for selecting sampling locations was therefore as follows:

- Identify and target a subset of discharges in each point source (and overland flow) category that can be used to develop representative loads for other discharges within the category that were not sampled during this effort.
- Identify and target unique discharges that may represent a potentially significant source to the Study Area.

Concentrations of TSS, TPAH (17), TPCB, and Cu measured during Phase 2 were used to establish representative concentrations (and ranges) for individual discharges or discharge types. These representative concentrations (and ranges) were then used to calculate a range of loads (i.e., arithmetic average, upper-, and lower-bound load estimates).

Four load calculation methods covering a range of data treatment methods were developed to quantify loads from sampled and unsampled discharges. A detailed discussion of the four loading methods is provided in Section 3.3 of Appendix E; a brief description is provided in this section and summarized in Table 5-11.

In all four methods, treated effluent from groundwater remediation and dewatering systems and WWTP treated effluent overflow loads were calculated by multiplying the annual discharge for each location by the range of concentrations (i.e., arithmetic average, minimum and maximum concentrations) of the three samples collected at each location. It was not necessary to estimate concentrations using the methods discussed in the following subsection, because all of these outfalls were sampled, whereas only a subset of the CSO and stormwater outfalls were sampled.[176]

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[176] The load from the groundwater extraction system at the Buckeye Pipeline Facility (DAR No. 106) was not estimated. Remedial pumping at the site operated from 1987 to October 2012. In March 2015, NYSDEC discontinued the SPDES permit due to the removal of the treatment system (GES 2017). This discharge was 0.22 MG per year, which is less than 0.01% of the total point sources discharge volume to the Study Area and less than 0.10% of the total treated effluent discharge volume. As discussed in the draft SSAM, DMR data and permit limits could serve as an additional LOE in the development of site-specific loading estimated for Category 1 treated groundwater effluent discharges when the site is required to monitor for Phase 2 analytes. This site monitored the discharge annually for pH; oil and grease; BTEX; and lead; therefore, TSS, TPCB, TPAH (17), and Cu loads could not be estimated.

For each of the four methods, stormwater[177] and CSO load estimates were added to the individual load estimates calculated for WWTP treated effluent overflow and treated effluent from groundwater remediation and dewatering system discharges described previously to complete the overall Study Area and reach-by-reach load estimates. This included upper- and lower-bound estimates of concentration data (i.e., for the various point source discharge types) to understand the range of possible loads to the system, and this is discussed in detail in Section 3.3 of Appendix E and is briefly summarized in Sections 5.1.4.2.1 through 5.1.4.2.4.

The four methods evaluated are referred to as the following: 1) the Collective Data Method (CDM); 2) the Specific Outfall Method (SOM); 3) the Representative and Unique (Stormwater) Method (RUM); and 4) the SOM/RUM hybrid method. These four methods span the range of possible data treatments with respect to combining data versus using individual measurements from sampled discharges. The range of load estimates explicitly accounts for the uncertainty and variability in the point source concentration measurements as discussed in Section 4.2 of Appendix E, so it will be used in future Newtown Creek RI/FS analyses, including CFT modeling.

5.1.4.2.1 Collective Data Method

All concentration data for CSOs were pooled, and for each analyte, the estimated mean[178] concentration and 95% UCL and lower confidence limit (LCL) of the mean were used to estimate the load for all sampled and unsampled outfalls. Likewise, all stormwater concentration data were pooled for each analyte, and the estimated mean concentration and 95% UCL and LCL were used to estimate the load for all sampled and unsampled outfalls.

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[177] Category 1 and Category 3 stormwater discharges were evaluated together based on information observed during reconnaissance and sampling that indicated the basin characteristics for these discharges were similar in many instances. For example, it was originally assumed that Category 1 discharges had a higher level of treatment than Category 3C discharges. However, it was discovered during reconnaissance that many Category 3C discharges also had stormwater treatment systems in operation, which provided support for combining these with Category 1 discharges.

[178] The term "mean" refers to the population mean (i.e., a parameter, the expected value of the population from which the data are sampled) and is distinct from the term "average," which refers to the sample mean (i.e., a statistic, the arithmetic average of the sample data).

5.1.4.2.2 Specific Outfall Method

At outfall locations that were sampled, for each analyte, location-specific minimum, maximum, and arithmetic average values were used to estimate the load for CSOs and stormwater. At unsampled outfall locations, the CDM (described in Section 5.1.4.2.1) was used to estimate the load for CSOs and stormwater.

5.1.4.2.3 Representative and Unique (Stormwater) Method

All concentration data for CSOs were pooled, and for each analyte, the estimated mean concentration and 95% UCL and LCL were used to estimate the load for all sampled and unsampled outfalls. At sampled and unsampled stormwater outfalls, concentration data from *a priori* representative locations (i.e., NCB629, NCQ632B, and NCB631) and locations not identified as statistically different (i.e., similar to the three representative locations; see Section 3.3.3 of Appendix E) were pooled. For each analyte, the estimated mean concentration and the 95% UCL and LCL of the pooled data were used to estimate the load.

5.1.4.2.4 RUM/SOM Hybrid

At outfall locations that were sampled, for each analyte, location specific minimum, maximum, and arithmetic average values were used to estimate the load for CSOs and stormwater. For unsampled CSO outfall locations, all CSO concentration data were pooled for each analyte, and the estimated mean concentration and 95% UCL and LCL were used to estimate the load. For unsampled stormwater outfall locations, concentration data from *a priori* representative locations (i.e., NCB629, NCQ632B, and NCB631) and locations not identified as statistically different (i.e., similar to the three representative locations) were pooled. For each analyte, the estimated mean concentration and the 95% UCL and LCL of the pooled data were used to estimate the load.

5.1.4.3 Results of Estimated Point Source Loads

For each reach of the Study Area, load estimates from each type of discharge were summed to calculate the total estimated load from point sources and overland flow[179] for the reach.

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[179] Point source discharges referred to in this section include discharges associated with overland flow, where present.

The total point source load to the Study Area was calculated by summing the reach-specific loads. Upper- and lower-bound estimates of the loads were calculated for each discharge category within each reach. As described in Section 3.3 of Appendix E, upper- and lower-bound load estimates were calculated from either minimum and maximum concentrations for locations that were treated individually (i.e., loads based on location-specific concentration and flow data) or from 95% LCLs and UCLs for cases where locations within a category were grouped (e.g., representative locations). It is important to note that summing the individual lower-bound estimates (or upper-bound estimates) for each discharge category within each reach to get an overall lower-bound (or upper-bound) load estimate to the Study Area assumes that all point source discharges are simultaneously at their low concentration (or high concentration), which is extremely unlikely. This is a conservative method for estimating the overall loads.

Loads for TSS, TPAH (17), TPCB, and Cu (including upper- and lower-bound estimates) for CSOs, stormwater, WWTP treated effluent overflow, and treated effluent from groundwater remediation and dewatering systems by reach and totaled across the Study Area for all four calculation methods are presented in Tables 5-12a through 5-15d and Figures 5-12 through 5-20. The tables present the annual flows, as well as concentrations and load estimates for TSS, TPAH (17), TPCB, and Cu associated with each load calculation method, with subtotals by reach and discharge type and totals across the Study Area. The figures present estimated loads by reach and discharge type for each of the load calculation methods. The results are discussed in Sections 5.1.4.3.1 through 5.1.4.3.5. As discussed in Section 5.1.4.2, the methods for estimating loads for WWTP treated effluent overflow and treated effluent from groundwater remediation and dewatering systems do not vary among load calculation methods.

5.1.4.3.1 TSS

The estimated TSS load for each method is summarized in Tables 5-12a through 5-12d and Figures 5-12 through 5-14. Total estimated TSS loads from point source discharges to the Study Area range from 840 to 1,000 metric tons per year (MT/year) across the four calculation methods, and the lower and upper bounds range from 610 to 1,500 MT/year (see Tables 5-12a through 5-12d). CSOs contribute the majority (52% to 63%) of the total point source TSS load, with other stormwater (19% to 25%) and MS4 (12% to 17%) discharges being the next largest contributors. WWTP treated effluent overflow and treated effluent from groundwater remediation and dewatering systems account for approximately 6% and less than 1%, respectively, of the point source TSS load to the Study Area.

East Branch, English Kills, and Maspeth Creek receive the highest TSS loads. TSS loads from point source discharges to CM 1 – 2 and CM 2+ are lower than those in these upstream tributaries (see Tables 5-12a through 5-12d and Figure 5-13). TSS loads to CM 0 – 1, Dutch Kills, and Whale Creek are the smallest loads to the Study Area.

Non-representative method calculations (i.e., CDM and SOM) result in higher TSS load estimates for the reaches in the main stem (see Tables 5-12a through 5-12d and Figure 5-13). TSS loads are slightly lower for SOM in Dutch Kills, East Branch, and English Kills, but they are slightly higher for SOM in Maspeth Creek. Overall, the estimated total TSS load to the Study Area is highest for the CDM and lowest for the RUM/SOM hybrid, but all methods are within 20% of each other (see Figure 5-12).

5.1.4.3.2 TPAH (17)

The estimated TPAH (17) load for each method is summarized in Tables 5-13a through 5-13d and Figures 5-12, 5-15, and 5-16. The estimated TPAH (17) load to the Study Area from point source discharges is estimated to be between 51 and 55 kg/year across the four calculation methods, with upper- and lower-bound estimates ranging from 2 kg/year to 99 kg/year, respectively (see Tables 5-13a through 5-13d). CSO discharges are the highest point source contributor of TPAH (17) to the Study Area (36% to 39%), followed by groundwater effluent (31% to 33%), other stormwater (15% to 18%), and MS4 discharges (10% to 12%). The WWTP treated effluent overflow contributes less than 2% of the estimated TPAH (17) load from point sources to the Study Area (see Tables 5-13a through 5-13d).

The load to CM 0 – 1 represents approximately one-third of the TPAH (17) load from point sources to the Study Area, which is largely a result of the estimated load from the Con Edison – 11th Street Conduit (DAR No. 110; CE11SC) (see Tables 5-13a through 5-13d and Figure 5-16). As described previously, groundwater that infiltrates into the Con Edison – 11th Street Conduit is pumped to an oil-water separator prior to being discharged to the creek. Elevated TPAH (17) concentrations in samples from that discharge (e.g., two or more orders of magnitude greater than any other point source samples; see Figure 5-6 and Table 5-5) account for the large calculated load, although it should be recognized that this value is uncertain due to lack of flow measurements (see Section 5.1.2).

In general, estimated TPAH (17) loads to East Branch, English Kills, and Maspeth Creek are slightly higher than the estimated loads to Dutch Kills, CM 1 – 2, and CM 2+, but are lower than the loads estimated for CM 0 – 1. The estimated TPAH (17) load to Whale Creek is smaller than the loads to other reaches (see Tables 5-13a through 5-13d and Figures 5-15 and 5-16).

For CM 1 – 2 and CM 2+, the TPAH (17) loads are slightly lower for the representative methods (i.e., RUM and RUM/SOM hybrid; see Table 5-13d and Figure 5-16). The SOM and RUM/SOM hybrid loads are higher in Dutch Kills and Maspeth Creek, but are lower in East Branch. The TPAH (17) load estimates to Whale Creek and English Kills are similar for the four methods. Likewise, the total TPAH (17) load estimates to the Study Area are nearly the same for all four methods (see Figure 5-12), because they are largely driven by the CE11SC load, which is calculated in the same way in all four methods.

5.1.4.3.3 TPCB

The estimated TPCB load for each method is summarized in Tables 5-14a through 5-14d and Figures 5-12, 5-17, and 5-18. The estimated TPCB load to the Study Area from point source discharges is estimated to be between 0.43 and 0.84 kg/year across the four calculation methods, and the lower and upper bounds range from 0.24 to 1.8 kg/year (see Tables 5-14a through 5-14d). The three largest current contributors of the point source TPCB load to the Study Area are CSO (22% to 49%), MS4 (21% to 28%), and other stormwater discharges (21% to 46%). Less than 12% of the estimated load comes from WWTP treated effluent overflow, and less than 0.1% comes from treated effluent from groundwater remediation and dewatering systems.

The highest TPCB loads from point sources occur in the upper reaches (i.e., CM 1 – 2, CM 2+, East Branch, English Kills, and Maspeth Creek) (see Tables 5-14a through 5-14d and

Figures 5-17 and 5-18). CM 0 – 1, Dutch Kills, and Whale Creek receive similar TPCB loads to each other, and these loads are smaller compared to the other reaches of the Study Area.

Non-representative method calculations (i.e., CDM and SOM) result in higher TPCB load estimates from point sources in the reaches of the main stem (i.e., CM 0 – 1, CM 1 – 2, and CM 2+) as compared to results from the representative methods (RUM and RUM/SOM hybrid; see Tables 5-14a through 5-14d and Figure 5-18). Estimated TPCB loads from point sources in the tributaries are lowest for the CDM and RUM in Dutch Kills but are highest in Maspeth Creek, East Branch, and English Kills. Overall, the total TPCB estimated point source load to the Study Area is highest for the CDM and lowest for the RUM/SOM hybrid, but all methods are within a factor or two.

5.1.4.3.4 Cu

The estimated Cu load for each method is summarized in Tables 5-15a through 5-15d and Figures 5-12, 5-19, and 5-20. The total estimated Cu load to the Study Area from point source discharges ranges from 390 to 460 kg/year across the four calculation methods, and the lower and upper bounds range from 280 to 660 kg/year (see Tables 5-15a through 5-15d). The three largest current contributors of the point sources Cu load to the Study Area are CSO (45% to 53%), MS4 (17% to 19%), and other stormwater discharges (20% to 26%). Less than 9% of the estimated load comes from WWTP treated effluent overflow, and less than 1% comes from treated effluent from groundwater remediation and dewatering systems.

East Branch, English Kills, and Maspeth Creek receive the highest Cu loads from point sources to the Study Area (see Tables 5-15a through 5-15d and Figure 5-19). Estimated Cu loads to CM 1 – 2 and CM 2+ are lower than estimated loads to East Branch, Maspeth Creek, and English Kills, but higher than Cu loads to Whale Creek and Dutch Kills. Estimated loads to CM 0 – 1 are the lowest of the reaches.

Non-representative method calculations (i.e., CDM and SOM) result in higher point source Cu loads for CM 0 – 1 and CM 2+ (see Tables 5-15a through 5-15d and Figure 5-19). In CM 1 – 2 and Whale Creek, the Cu loads from all four methods are approximately the same. The Cu loads are lower for the SOM and RUM/SOM hybrid in Dutch Kills, East Branch, and

English Kills, but are higher in Maspeth Creek as compared to the CDM and RUM. Overall, the total Cu estimated load from point sources to the Study Area is highest for the CDM and lowest for the RUM/SOM hybrid, but all methods are within 20% of each other (see Figure 5-12).

5.1.4.3.5 Load Estimates Summary

Comparisons of the four different methods of load estimation indicate that results are similar to one another. The load estimates are influenced more by discharge volumes and, to some extent, by spatial distribution of chemical concentrations; they are generally not sensitive to treating individual and unique locations differently from a more generalized or representative approach. For TSS, TPAH (17), TPCB, and Cu, the four load methods produced estimates that are within a factor of 3 for each reach, and when compared across discharge types (e.g., CSO loads calculated by the four loading methods), the estimates are generally within a factor of 2 for each discharge type (see Figures 5-13, 5-15, 5-17, and 5-19). The load estimates from the four different methods are within a factor of 2 for the Study Area as a whole (see Figure 5-12). In general, the highest loads for the Study Area are calculated using CDM, and the lowest loads are calculated using the RUM/SOM hybrid. As such, these two methods are used to establish a representative range for comparison purposes in the mass balance discussion in Section 6.5 and in the CSM discussion in Section 8. In this analysis, point source loads are further evaluated by comparing them to loads from other sources (e.g., from groundwater), estimates of fluxes between sediment and water, and the mass inventory within surface water and surface sediment. Additionally, ranges of load estimates for point sources will be further evaluated with the CFT model being developed for the FS, which will include development of uncertainty bounds for modeling sensitivity/uncertainty analyses.

5.1.5 Evaluation of Variability and Potential Uncertainties in the Point Source Load Estimates

The results of the point source load analyses presented in this section and in Section 3.3 of Appendix E meet the objectives laid out in the RI/FS Work Plan (AECOM 2011) and provide the data needed to support a quantitative evaluation of the relative current contributions of the point sources for RI/FS purposes. Point source load calculations are dependent on two types of data—flow and concentration; variability and potential uncertainties associated with each of these types of data are discussed in detail in Section 4 of Appendix E. The primary uncertainties in the point source load estimates are related to the inherent variability of stormwater and CSO discharges (e.g., due to variations in annual precipitation, proportion of sewage in CSO discharges) relative to the number of samples collected and the frequency of sample collection. Annual loads for a given year will vary based on actual precipitation amounts and associated point source discharges. To address variability in the flow term, future load evaluations conducted with the CFT model will estimate loads for each year of the simulation driven by precipitation data from the meteorological record. The range of load estimates explicitly accounts for the uncertainty in the concentration term and will thus be carried forward into the CFT modeling.

5.2 Groundwater

Groundwater is a potential ongoing source of chemicals to the Study Area. Groundwater discharge to the Study Area consists of discharge to the base of the Study Area and lateral groundwater flow through vertical permeable shorelines[180] to the surface water (i.e., lateral discharge). Groundwater discharge to the base of the Study Area may provide chemical loads to the subsurface sediment via transport through the interstitial spaces (as porewater) and sorption onto the solid matrix. However, due to attenuation within the subsurface sediment, the chemical loads associated with groundwater only indirectly contribute to the chemical loads to the biologically active surface sediment and surface water (as discussed further in Section 6.4.5; this process is being further evaluated with the CFT modeling being developed to support the FS). The estimated annual loads of TPAH (17), TPCB, and Cu from groundwater to subsurface sediment were calculated as the product of groundwater discharge (i.e., flow rate) to the base of the Study Area and dissolved phase chemical concentrations present within groundwater beneath the Study Area. Groundwater discharge to the base of the Study Area is interpolated from seepage meter measurements obtained at the sediment surface, as well as seepage rates calculated based on vertical hydraulic gradient measurements and vertical hydraulic conductivity estimates.

Chemical loads associated with lateral discharge were not quantified for this RI Report, because shallow lateral groundwater discharge rates to the Study Area were not empirically

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[180] Vertical permeable shorelines include pile-supported concrete, precast concrete blocks, and vertical wood bulkheads.

measured and chemical concentrations in the upland fill unit groundwater are not known. Instead, a qualitative assessment of surface water quality in areas of relatively high lateral discharge has been performed (see Section 5.2.2.4) to understand if lateral groundwater discharges have a notable effect on surface water quality. During the development and calibration of the CFT model, chemical loads from lateral discharge will be further evaluated through the completion of sensitivity analyses. In addition, USEPA is planning a study to further characterize shallow lateral groundwater discharge along the shoreline of Newtown Creek. The stated objective of the USEPA study is to collect empirical data to achieve sufficient characterization of shallow lateral groundwater discharge to support the FS and reduce uncertainty in the current lateral groundwater discharge estimates.

Due to the long history of industrial development, urbanization, and discharges into the Study Area from numerous uncontrolled sources (see discussions in Section 3) and the dynamic transport and depositional processes within the Study Area, the locations of elevated groundwater TPAH (17), TPCB, and Cu concentrations and loads cannot be definitively linked to proximate upland sites. Nevertheless, groundwater samples collected during the RI provide a snapshot of the present-day distribution and concentrations of TPAH (17), TPCB, and Cu that contribute current chemical loads from groundwater to the base of the Study Area sediment.

The estimation of groundwater flow and associated TPAH (17), TPCB, and Cu loads to the Study Area follows the approaches outlined in the *Final Groundwater Investigation Work Plan* (USEPA 2014a) and the MAM2 (Anchor QEA 2016c). Flow and chemical concentration data collected during the Phase 2 groundwater and FS Part 1 sampling programs are presented in Appendices Bi and Bii. Flow data are discussed in detail in Sections 3.5 and 5.2.1 of Appendix F and are summarized in this section. The distribution of dissolved phase TPAH (17), TPCB, and Cu in groundwater is described in Section 4.9.3, with additional discussion in Section 3.7.2 of Appendix F. Load calculations and supporting information are presented in Section 6 of Appendix F. Calculated loads for TPAH (17), TPCB, and Cu are summarized in this section.

As outlined in the MAM2 and described in Appendix F (see Section 1.2), estimation of groundwater discharge for the RI follows a three-tiered approach, in which each tier presents more detail than the preceding tier (see Figure 5-21). The approach and results for each tier are presented in Section 5 of Appendix F. The Tier 1 analysis is a regional water balance for the water-bearing units within the potential groundwater contribution area, within which the groundwater that discharges to the Study Area likely originates. Tier 1 analysis accounts for groundwater inflow from recharge and artificial returns (e.g., leakage from waterlines and sewers), as well as groundwater outflow (e.g., groundwater withdrawal, dewatering, and groundwater loss to sewer pipes). Discharges of treated[181] groundwater effluent to Newtown Creek, which result from groundwater withdrawals, are accounted for in the hydrodynamic model as inflow terms (see Section 4.4.4 of Appendix G). In Tier 2, groundwater discharge to the Study Area is estimated based on measured and calculated groundwater seepage rates, which are interpolated across the Study Area. In Tier 3, groundwater discharge is evaluated for each of 56 shoreline segments, which were identified based on several physical attributes, such as hydraulic gradients, upland sites of interest, shoreline structures, and sediment thickness. The 56 segments comprise 13 segment groups. Segment groups are shown in Figure 5-22, and individual segments are shown in Figures 5-23a through 5-23c. The Tier 1 analysis is described in Section 5.1 of Appendix F. The approaches for the Tier 2 and Tier 3 evaluations are presented in Sections 5.2.1.1 and 5.2.1.2.

5.2.1 Groundwater Discharge

5.2.1.1 Tier 2 – Interpolated Seepage Rates

Groundwater flow and chemical load estimates will ultimately be used as a boundary condition for the CFT model being developed for the Study Area; therefore, the calculation of groundwater discharge rates was developed at the same spatial scale as the hydrodynamic and CFT model grid (described in detail in Section 5.2 of Appendix F).

In Tier 2, groundwater seepage rates for 38 locations were interpolated for each grid cell using a triangular irregular network surface. The seepage dataset includes 15 Phase 2 seepage rate measurements, 13 FS Part 1 seepage rate measurements, 6 calculated seepage rates at Phase 2 in-creek long-term monitoring stations, and 4 seepage rates calculated using FS Part 1 vertical hydraulic gradient data. The 15 Phase 2 and 13 FS Part 1 seepage rate measurements were collected by USGS under contract to USEPA; these data quantify the net

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[181] With regard to "treated groundwater effluent," the degree of treatment is variable and documented in each individual SPDES permit.

flow of water across the sediment-surface water interface (i.e., mudline) and were used to estimate the net groundwater discharge to the base of the Study Area, as described in the USEPA's *Final Groundwater Investigation Work Plan* (USEPA 2014a). The two USGS reports documenting seepage meter measurements are included in Attachments Bi-B9 and Bii-B1 of Appendices Bi and Bii, respectively. Seepage rates at the six in-creek long-term monitoring stations were calculated using the arithmetic average vertical hydraulic gradient, based on long-term potentiometric level data collected by Anchor QEA during Phase 2 and hydraulic conductivity values from Phase 2 slug tests. In consultation with USGS, Anchor QEA calculated seepage rates at the four FS Part 1 vertical hydraulic gradient measurement locations using the arithmetic average vertical hydraulic gradient and the arithmetic average vertical hydraulic conductivity for the Study Area sediments (see Section 5.2.1 of Appendix F for details). The basis of the interpolation of groundwater seepage rates is the net seepage rates derived from seepage meter deployments or vertical hydraulic gradient measurements over several tidal cycles (including months of deployment in the case of long-term potentiometric level data). At many locations, the gross seepage rates or vertical hydraulic gradients vary between upward and downward directions in response to tides.

Figures 5-24a through 5-24c show the interpolated seepage rates within the Study Area. The highest positive seepage rates generally occur at or near the head of English Kills (up to 8.3 centimeters per day [cm/day]), at CM 2.5 to 2.7 (up to 8.2 cm/day), and at CM 2.0 to 2.1 (up to 3.7 cm/day). Seepage rates are generally positive (i.e., upward from sediment into surface water), except in the following areas:

- The central portion of Newtown Creek between CM 1.5 and 2.0
- Just upstream of the mouth of Dutch Kills near CM 1.1
- Just downstream of the Pulaski Bridge near CM 0.5

Net negative seepage indicates a net loss of water from the creek to groundwater (i.e., downward flow). Net negative seepage at CM 1.5 to 2.0 (up to -7.4 cm/day) is attributed to groundwater extraction at a total rate of 670 gallons per minute at adjacent upland groundwater remediation sites.[182] The groundwater extraction rates at individual upland remediation sites and the references documenting the extraction rates are described in Sections 4.5.2.1 and 5.1.2.1 of Appendix F.

The specific cause of slightly negative net seepage near CM 1.1 (-0.3 cm/day) is unknown, though it is likely due to an extraction system or systems. Slightly negative net seepage near CM 0.5 (-0.3 cm/day) may relate to known dewatering of a utility tunnel located in that area.

The Tier 2 volumetric groundwater discharge rate for each model grid cell is calculated by multiplying the Tier 2 interpolated seepage rate by the surface area of the grid cell.

5.2.1.2 Tier 3 – Groundwater Discharge for Individual Segments and Segment Groups

Total groundwater discharge to the Study Area is calculated by summing the discharge to the base of the Study Area and lateral discharge, which is estimated here as a percentage of the groundwater discharge to the base of the Study Area within corresponding segments. As previously stated, shallow lateral groundwater discharge has not been empirically characterized and USEPA is planning a study to collect empirical data with the stated objective of reducing uncertainty in the current lateral groundwater discharge estimates as part of the FS.

Groundwater discharge to the base of the Study Area for the 56 segments is estimated by summing the Tier 2 discharge rates for all the grid cells within each segment and then making adjustments, as appropriate, based on the estimated hydraulic effects of select shoreline structures bordering each individual segment. Shoreline types vary throughout the Study Area and were simplified and combined into the following three primary categories for the Tier 3 analysis: permeable shorelines, shallow barriers, and deep barriers. Each continuous section of a single shoreline category within a given segment is a subsegment. Each primary category of shoreline has a different hydraulic effect on groundwater flow. Permeable shorelines are assumed to be fully permeable to groundwater flow (i.e., 100% of flow passes through).

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[182] It is recognized that there is potential for the existence of small, localized areas between CM 1.5 and 2.0 where groundwater discharges to the Study Area (net positive seepage), based on the potentiometric contours for the UGA shown in Figure F3-11 of Appendix F.

Shallow barriers and deep barriers are assumed to reduce the discharge to (or from) a segment or subsegment by 5% and 50%, respectively. The flow that is blocked by shallow barriers or deep barriers is assumed to commensurately increase groundwater discharge at adjacent, more permeable shorelines. The sensitivity of the groundwater discharge estimate to these assumed flow diversion percentages (e.g., 5% and 50%) is explored in the sensitivity analysis (see Section 5.2.3). The three shoreline categories are shown in Figures 5-24a through 5-24c. Shoreline surveys, characterization of shoreline structures, and categorization of shorelines are described in Section 3.2.2 of Appendix F. Details regarding the hydraulic effects of shoreline categories on groundwater discharge are provided in Section 5.3 of Appendix F. Lateral discharge for each segment that contains vertical permeable shorelines is estimated as a percentage of groundwater discharge to the base of the Study Area within the segment, as described in detail in Section 5.3.2 of Appendix F.

The estimated groundwater discharge rates for the 56 segments are presented in Table 5-16. The total groundwater discharge to the Study Area is 250,000 cubic feet per day (feet^3/day; 1.8 MGD), of which 190,000 feet^3/day (1.4 MGD) is discharge to the base of the Study Area and 54,000 feet^3/day (0.40 MGD) is lateral discharge through vertical permeable shorelines. Table 5-16 shows the multiplier for each segment, which is the ratio of the Tier 3 discharge rate for a given segment to the Tier 2 discharge rate for the same segment (due to adjustments based on shoreline structures). Figure 5-25 shows the resulting Tier 3 modified seepage distribution for model grid cells, which was obtained by multiplying the Tier 2 interpolated seepage rates within an individual segment by the multiplier for that segment. The Tier 3 modified seepage is similar to the Tier 2 interpolated seepage in Figures 5-24a through 5-24c. The Tier 3 modified seepage rates shown in Figure 5-25 were used as the groundwater discharge component for estimating the chemical loads presented in Section 5.2.2. Table 5-16 also shows the estimated lateral discharge and the lateral groundwater discharge rate per linear foot of shoreline for each segment.

As a check on Tier 3 volumetric discharge rates, groundwater discharge estimates for the 13 segment groups were calculated by summing the total Tier 3 volumetric flow rates (including discharge to the base of the Study Area and lateral discharge) for model grid cells aligned with each segment group and using those results to back-calculate upland hydraulic properties and net recharge rates. Details regarding this analysis are provided in Section 5.3.3 of Appendix F.

The resulting hydraulic conductivity and recharge values from this back-calculation are highly variable, but this is not unexpected based on the range of reported hydraulic conductivity values and the uncertainties underlying the regional water balance. As shown in Figure F3-16 and Tables F3-9, F3-12, F3-14, and F5-14 of Appendix F, measured hydraulic conductivity values for the UGA vary by orders of magnitude, but the arithmetic average Tier 3 calculated hydraulic conductivity is similar to that from other studies (see Table F5-14). Due to the following factors, the transmissivity and hydraulic conductivity values estimated based on Tier 3 can be expected to differ from one segment group to the next:

- The complexity and heterogeneity of the moraine deposits that comprise the UGA in the vicinity of the Study Area
- The large range in measured hydraulic conductivity values for the UGA reported in the literature
- Differences in the number and magnitude of potential groundwater sources and sinks within each segment group
- Differences in the number and locations of available hydraulic head data that were used to estimate the hydraulic gradients and groundwater flow directions within the upland

Differences in back-calculated recharge rates from one segment group to the next are also expected given the complex environment, including geology, land cover, and infrastructure. In addition, the back-calculated equivalent recharge rates include recharge from precipitation, artificial returns, losses to sewer pipes, and dewatering, all of which can differ greatly from segment group to segment group and within each segment group. While the estimated sizes and shapes of upland segment group areas and the groundwater discharge areas in the creek assigned to each segment group were determined based on the available upland hydraulic head data, they should not be considered precise. For example, the relatively low estimated net recharge for Segment Group F is the result of low groundwater discharge assigned to this segment group and the relatively large upland area estimated for this segment group. The relatively high estimated net recharge for Segment Group J results from the relatively high groundwater discharge assigned to this segment group and the relatively small upland area estimated for this segment group. Also, cross-sectional modeling results presented in Attachment F-K of Appendix F indicate that the divide between groundwater discharges assigned to opposing sides of the creek may not be at the center of the creek in all areas, as assumed in the equivalent recharge rate calculations generated from the Tier 3 groundwater model. For example, in Cross-Sectional Model No. 2, the hydraulic divide beneath the creek was predicted to be closer to the Brooklyn side than the Queens side of the Turning Basin. Therefore, the back-calculated recharge rate may be overestimated for the Brooklyn side (Segment Group J) and underestimated on the Queens side (Segment Group F). All of these factors contribute to the differences in back-calculated equivalent recharge rates between segment groups. Nevertheless, the calculated net recharge rates are within the expected range based on Tier 1 recharge estimates and published estimates (see Section 5.1 of Appendix F).

5.2.1.3 Comparison of Groundwater Seepage to Vertical Salinity Profiles

Salinity data may provide a qualitative indicator of the direction of flow between groundwater and surface water in some circumstances. Vertical profiles of salinity from surface water, sediment porewater, and groundwater at 17 locations (Figures 4-231a and 4-231b) were compared with estimated seepage rates at the same locations. All of these locations have net positive (upward) seepage; salinity vertical profiles are not available for any of the locations where net negative seepage was measured or calculated because of the absence of shallow and mid-depth porewater data at these locations.

Salinity vertical profiles at four of the five locations on the main stem between CM 0 and 1.3 do not show any notable differences in salinity concentrations, with groundwater salinity being within 1 psu of shallow porewater salinity and within 5 psu of surface water salinity (Figures 4-144 to 4-147b and Appendix F, Figure F3-21a). At the fifth location (NC266), the salinity in groundwater is slightly lower than salinity in shallow porewater (within 3 psu) and surface water (within 6 psu). The estimated seepage rates at these five locations range from 1.3 to 2.1 cm/day (upward).

Salinity vertical profiles at the other 13 locations in the tributaries and the main stem upstream from CM 1.4 (inclusive) generally show lower salinity in groundwater than in surface water and shallow porewater (with the exception of Location DK052) (Figures 4-144 to 4-147b and Appendix F, Figures F3-21a and F3-21b). The estimated seepage rates at these 13 locations range from 0 to 8.3 cm/day (upward). Salinity vertical profiles in these portions of the Study Area do not appear to differ as a function of groundwater discharge rate. For example, the shape and slope of the salinity vertical profiles for Locations NC286, NC062, and NC218 are similar (i.e., salinity in groundwater is 10 to 15 psu lower than salinity in shallow porewater and surface water), but the net groundwater discharge rates differ notably (0.0, 3.7, and 8.2 cm/day, respectively).

Overall, although the available data generally show lower salinity in groundwater than in shallow porewater and surface water at locations upstream of CM 1.4 (inclusive of the tributaries), there is no indication of any systematic relationship between salinity vertical profile and groundwater discharge rate. At all of these locations for which vertical salinity profile data are available, the net groundwater discharge is positive (flow toward the creek). Differences between salinity vertical profiles appear to relate mostly to proximity to the East River, with relatively high groundwater salinity near the East River (the average for CM 0 – 1 is 19 psu) and fresher groundwater in the upstream reaches of the Study Area (the average salinities measured in groundwater beneath CM 2+, English Kills, and East Branch are 4.4 psu, 7.2 psu, and 4.5 psu, respectively [Appendix F, Table F3-21 and Figures F3-19 and F3-20]).

As noted above, salinity vertical profiles are not available for any of the locations where net negative seepage was measured or calculated. However, in these areas, mixing would be expected between surface water and less saline groundwater. Mechanisms that may mix saline surface water with fresh groundwater in areas of either positive or negative groundwater discharge include tidal exchange, molecular diffusion, and mechanical dispersion (see Section 5.2.1.3 of Appendix F). These mechanisms occur regardless of the direction of the net groundwater seepage. In other words, where net groundwater seepage is downward, groundwater may still flow upward during low tide, reducing groundwater salinity below that of the surface water and porewater.

5.2.2 Estimated Groundwater Chemical Loads

Groundwater chemical load was calculated as the advective chemical flux from groundwater to subsurface sediment within the Study Area. Loads are calculated for TPAH (17), TPCB, and Cu in each segment with a net positive seepage rate based on the estimated Tier 3 groundwater discharge (presented in the previous subsection) and the dissolved phase concentrations of TPAH (17), TPCB, and Cu in groundwater in native material beneath the Study Area (discussed in Section 4.9.3). For segments with net negative seepage, the net chemical loads from groundwater to the Study Area are considered to be zero.[183]

Chemical loads from groundwater to the subsurface sediment within the Study Area were calculated using Equation 5-2:

$$W = Q \times C \tag{5-2}$$

where:

W = chemical load (M/T)

Q = groundwater discharge rate (L^3/T)

C = dissolved phase TPAH (17), TPCB, and Cu concentration in groundwater (M/L^3)

Dissolved phase chemical concentrations in groundwater samples collected from native material during the Phase 2 groundwater investigation (see Section 4.9) were used for the C term in Equation 5-2. As discussed in Section 4.9.1, dissolved Cu concentrations were measured directly in filtered samples, whereas dissolved TPAH (17) and TPCB concentrations were estimated from whole-water concentrations using partitioning calculations that account for the DOC-bound fraction. As discussed in Section 4.9.3, dissolved phase concentrations of TPAH (17) and TPCB in groundwater were estimated based on partitioning calculations using two different methods to bracket the range of uncertainty—Method 1, which used site-specific dry weight partition coefficients along with sample-specific measured TSS and DOC; and

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[183] In cases where areas with net negative seepage include times of gross positive seepage due to tidal fluctuations, the load associated with that process is quantified as part of the porewater exchange process (see Section 6.4.3) through use of a mass transfer coefficient that subsumes the magnitude of gross fluctuations about the net value (see Section 6.5.1.3).

Method 2, which used literature-based OC partition coefficients along with sample-specific measured TOC and DOC, or TOC from collocated native material samples when TOC measured in a groundwater sample was non-detect (see Section 4.8.3). Therefore, the ranges of TPAH (17) and TPCB concentrations from these two methods are used to develop a range of estimated loads from groundwater to the Study Area.[184] For each segment where more than one groundwater sample was collected, the arithmetic average of the dissolved phase chemical concentrations in the samples collected in that segment was used in the chemical load calculation. For segments where groundwater samples were not collected, arithmetic average dissolved phase chemical concentrations measured at nearby, adjacent sample locations were used. In most cases, the two closest sample locations—one at each end of the segment—were averaged and used for that segment.

The estimated loads of TPAH (17), TPCB, and Cu from groundwater to subsurface sediment in the Study Area are summarized by reach (i.e., CM 0 – 1, CM 1 – 2, CM 2+, and the five tributaries) in Tables 5-17a and 5-17b (TPAH [17] and TPCB only). The spatial distributions of TPAH (17), TPCB, and Cu loads from groundwater to subsurface sediment in the Study Area are shown in Figures 5-26a through 5-28. Sensitivity analyses for these estimates are discussed in Section 5.2.3.

5.2.2.1 TPAH (17)

The estimated dissolved phase TPAH (17) load from groundwater is between 830 and 1,500 kg/year. Estimated dissolved phase TPAH (17) loading is the highest along the Brooklyn shoreline of Newtown Creek, just downstream of the confluence with English Kills in the Turning Basin (CM 2.3 to 2.7). The estimated dissolved phase TPAH (17) load to CM 2+, which contains this area, represents 89% to 91% of the estimated dissolved phase TPAH (17) load from groundwater to subsurface sediment. The estimated dissolved phase TPAH (17) load is also elevated along the Queens shoreline of Newtown Creek, just downstream of the confluence with Dutch Kills between CM 0.6 and 0.9, and near the mouth of the creek between CM 0.0 and 0.4. The estimated dissolved phase TPAH (17) load to subsurface sediment from groundwater in

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[184] As presented in Section 3.7.2.1 of Appendix F, Method 2 may be more appropriate for estimating dissolved-phase concentrations in groundwater, because of the difference in OC content between native material and Study Area sediments. However, because both methods are uncertain, concentrations derived from both methods were used to bracket estimated TPAH (17) and TPCB loads to the base of the Study Area in the RI Report.

CM 0 – 1, which contains this area, represents 7% to 10% of the dissolved phase TPAH (17) load to subsurface sediment, based on Methods 1 and 2. However, as previously noted in Section 4.9.3 (and Section 6 of Appendix F), the estimated dissolved phase TPAH (17) loads from groundwater may be overestimated due to the possible presence of NAPL in a few groundwater samples with relatively high calculated dissolved phase TPAH (17) values. Estimated dissolved phase TPAH (17) loads to other reaches of Newtown Creek and the tributaries are much smaller, comprising the remaining 1% of the dissolved phase TPAH (17) load to subsurface sediment.

5.2.2.2 TPCB

The estimated dissolved phase TPCB load from groundwater to subsurface sediment is between 0.045 and 0.27 kg/year. Estimated dissolved phase TPCB loads are highest in the upstream reaches of English Kills and at discrete locations in East Branch and the upper portion of the Turning Basin. The estimated dissolved phase TPCB load to English Kills represents 81% to 90% of the total TPCB load from groundwater to subsurface sediment. Estimated dissolved phase TPCB loads from groundwater to subsurface sediment in the other reaches of Newtown Creek and the tributaries are at least one order of magnitude lower.

5.2.2.3 Cu

The estimated dissolved Cu load from groundwater is 10 kg/year. Dissolved Cu loads are highest in the Turning Basin (CM 2.0 to 2.8), in the upstream reaches of English Kills, and along the Queens shore of Newtown Creek between CM 1.1 and 1.5. The loads to the reaches containing these three areas represent 35% (CM 2+), 23% (English Kills), and 20% (CM 1 – 2) of the total load, respectively. Dissolved Cu loads are also elevated at discrete locations along the other reaches of Newtown Creek and the tributaries.

5.2.2.4 Qualitative Evaluation of the Influence of Lateral Groundwater Discharge on Surface Water Quality

To evaluate the potential influence of lateral groundwater discharge on surface water quality within the Study Area, dry weather surface water data (collected at different tide stages) adjacent to the five groundwater segments with the highest estimated lateral groundwater discharge rates (per linear foot of shoreline; see Table 5-16) were compared to those in surrounding areas of the creek. The comparison suggests no observable influence from lateral groundwater discharge on surface water chemical concentrations (see Section 6.4 of Appendix F). Nonetheless, definitive conclusions cannot be made based solely on this analysis. Therefore, the potential contribution of chemical loads associated with lateral groundwater discharge will be evaluated more quantitatively using the CFT model (i.e., through diagnostic analyses) being developed to support the FS (see Section 6.4.1); it will be documented in the modeling report and evaluated, as necessary, in the FS. Furthermore, as stated in Section 5.2, USEPA is planning a study to further characterize shallow lateral groundwater discharge along the shoreline of Newtown Creek.

5.2.3 Sensitivity Analysis

Sensitivity analyses were performed to assess possible ranges of groundwater discharge and chemical load estimates based on parameter uncertainty. Sensitivity analyses were performed for the Tier 1 water balance, Tier 2 groundwater discharge, Tier 3 groundwater discharge, and chemical loads. Details of the sensitivity analysis are described in Section 7 of Appendix F.

The sensitivity analysis results suggest the following:

- The Tier 1 water balance analysis has relatively high uncertainty, and the results are sensitive to changes in recharge from precipitation, artificial returns, and loss to sewer pipes. However, within a reasonable range of adjustments for these three parameters, the Tier 1 results are consistent with the Tier 3 results for total groundwater discharge to the Study Area.
- Using the minimum and maximum median seepage rates from the two seepage meter deployments (i.e., Phase 2 and FS Part 1 studies; see Section 5.2.1.1) at Stations EK013SP and EK098SP in English Kills (see Section 7.2 and Table F3-8 of Appendix F), the total Tier 2 groundwater discharge to the base of the Study Area in English Kills changes to 83% and 126% of the base case estimate, respectively.
- Adjusting the percent of flow diversion associated with deep barriers and shallow barriers in the Tier 3 model has negligible effect on the calculated groundwater discharge or estimated chemical load. In many cases the flow that is blocked by one or more flow barriers within a given subsegment is diverted to an adjacent permeable shoreline subsegment within the same segment.

- The main source of uncertainty for chemical load is the estimated dissolved phase concentrations for TPAH (17) and TPCB. In the Base Case – Lower Bound scenario, site-specific dry weight partition coefficients were developed from paired porewater and bulk sediment sample chemistry data, as described in Section 4.9.3 (i.e., Method 1; see also Section 3.7.2.1 of Appendix F). In the Base Case – Upper Bound scenario, literature-based OC partition coefficients were used to calculate dissolved phase concentrations (instead of the site-specific dry weight values used in the base case; this is Method 2). Method 2 results in estimated chemical loads that are a factor of 2 to 6 times larger than loads derived from Base Case – Lower Bound, depending on the chemical. In addition, potentially entrained NAPL in groundwater samples collected in a few areas likely elevated the apparent dissolved phase concentrations of TPAH (17).

5.3 East River

Due to the twice-daily tides, the East River transports solids into the Study Area that contain contaminant concentrations reflecting the regional urban setting. An evaluation of solids entering Newtown Creek from the East River, including quantification of loads, is provided by the sediment transport model (see Section 5.6.1.2 of Appendix G). To understand concentrations of chemicals in surface water entering Newtown Creek from the East River, surface water samples were collected monthly for 10 months from the East River (at Location ER001, which is approximately 0.2 mile west of the mouth of Newtown Creek) and from three locations along a transect at the mouth of Newtown Creek. Figure 5-29 shows the locations where surface water samples were collected for the East River program. Samples were collected under flood tide conditions for 10 months; for 3 months of the sampling program, surface water samples were also collected from the transect locations under an ebb tide condition. Although samples were collected under both tide conditions, flow patterns at the mouth of Newtown Creek at the location of the mouth transect are complex. Small-scale current patterns could result in differing flow directions than anticipated at the prevailing tide at one or more of the three locations that make up the transect. These small-scale current patterns have been observed when inspecting outputs from the hydrodynamic model that is described in Section 4 of Appendix G. Due to these complexities, all data from the sampling transect at the mouth, regardless of tide condition, were combined for the purposes of the discussions in this section.[185]

Data collected during the Phase 2 East River surface water sampling program are presented in Section 3 of Appendix Bi.[186] Results for OC, TSS, TPAH (17), TPCB, and Cu are discussed in this section. Based on the calculations summarized in Section 4.1.3.5 and outlined in detail in Attachment E-C of Appendix E, estimated particulate phase concentrations were calculated for the East River surface water sampling program; these are also described in this section.

The complex circulation pattern at the mouth of the creek also complicates quantification of contaminant loads entering the Study Area from the East River. As discussed in Section 6.5, due to this complexity, empirical-based loads cannot be reliably calculated. However, these loads will be quantified as part of the CFT modeling being performed as part of the FS. The modeling will make use of the data described in this section.

5.3.1 Organic Carbon

For the East River sampling, POC and DOC FoD was low, at 15% and 7%, respectively.[187] These low FoD values preclude any attempt to discern patterns in these data. The TOC FoD was 43% in the East River surface water samples. As a result, some observations can be made from that dataset. The arithmetic average of TOC concentrations in surface water under flood tide conditions from ER001 was 5.4 mg/L, with a range of 4.6 to 8.5 mg/L. TOC concentrations in surface water from the mouth transect locations under flood tide conditions were similar to ER001. Summary statistics for concentrations measured in the East River sampling program are included in Table 5-18.

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[185] The data from both tide conditions were combined in the figures presented in this section and are discussed together in the text of this section. However, they are listed separately in the tables presented in this section as a means of illustrating that the differences between samples collected under flood and ebb tide conditions were small (recognizing there were far fewer samples collected during ebb tide conditions).

[186] Tables of East River surface water concentration summary statistics are provided for the full list of chemical analytes in Appendix A-B. These analytes include TPAH (34), 2,3,7,8-TCDD, and Pb; East River surface water samples were not analyzed for C19-C36.

[187] The POC and DOC MDLs for the East River program were similar to those for the other RI surface water programs.

Figure 5-30[188] shows the temporal pattern of TOC concentration of each sample from the East River sampling program. When detected, TOC concentrations in surface water from the mouth transect locations under flood tide conditions are similar to ER001, with the exception of some elevated TOC concentrations in ER001 during the November 2014 sampling event. Concentrations are the highest in the September sampling event relative to November through February, which could suggest a seasonal relationship, although the detected data are too limited to draw a definitive conclusion.

5.3.2 TSS

East River TSS concentrations in surface water under flood tide conditions from ER001 had an arithmetic average of 28 mg/L, with a range of 10 to 45 mg/L. TSS concentrations in surface water from the mouth transect locations under flood tide conditions were similar to ER001. Summary statistics for concentrations measured in the East River sampling program are included in Table 5-19.

Figure 5-31 shows the temporal pattern of TSS concentration of each sample from the East River sampling program. TSS concentrations are similar in samples from the East River and the mouth transect locations. There is no apparent relationship between TSS concentration and sample depth, indicating that the water column is vertically well mixed at these sampling locations. There is no apparent seasonal pattern in TSS concentration in the East River surface water dataset. The months with the highest TSS concentrations are July (a warm month) and February (a cold month). Within a given month's sampling event, concentrations do not vary greatly, suggesting that month-to-month variations in TSS are linked to factors other than temperature or time of year.

Additional evaluation of TSS in the East River was conducted as part of the sediment transport modeling, as discussed in Section 5.4.3.1 of Appendix G. These evaluations include a June 2018 sampling program conducted under Part 1 of the FS to characterize the grain size distribution of East River solids, as well as quantification of the loads of solids entering the Study Area from the East River.

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[188] The data from samples collected under both flood and ebb tide conditions were combined in this and all other figures in this section (see Section 5.3 and Footnote 185).

5.3.3 TPAH (17)

TPAH (17) concentrations in surface water under flood tide conditions from ER001 had an arithmetic average of 0.15 µg/L, a median of 0.13 µg/L, and a range of 0.050 to 0.47 µg/L. TPAH (17) concentrations in surface water from the mouth transect locations under flood tide conditions were similar to ER001. Summary statistics for concentrations measured in the East River sampling program are included in Table 5-20.

Figure 5-32 shows the temporal pattern of TPAH (17) concentration of each sample from the East River sampling program. TPAH (17) concentrations are similar in samples from the East River and the mouth transect locations. There is no apparent relationship between TPAH (17) concentration and sample depth, indicating that the water column is well mixed. Concentrations were slightly higher during some of the warmer months, with the highest (and most variable) concentrations measured occurring in August; differences in monthly arithmetic averages are generally less than a factor of 2 to 3.

Estimated particulate phase TPAH (17) concentrations in the East River under flood tide conditions from ER001 had an arithmetic average of 3.2 mg/kg, a median of 3.2 mg/kg, and a range of 0.75 to 7.2 mg/kg. TPAH (17) concentrations in surface water from the transect locations under flood tide conditions were similar to ER001. Summary statistics for concentrations measured in the East River sampling program are included in Table 5-21.

Figure 5-33 is a box plot of the estimated particulate phase surface water concentrations in samples from the East River program, the Study Area, and Phase 2 reference areas during dry weather. Particulate phase concentrations are similar in the East River and throughout most of the Study Area. Concentrations in the East River are slightly greater than in the Phase 2 reference areas, with the exception of Westchester Creek, where they are similar.

5.3.4 TPCB

TPCB concentrations (all samples were analyzed using congener-specific PCB methods) in surface water under flood tide conditions from ER001 had an arithmetic average of 4.8 ng/L, a median of 4.0 ng/L, and a range of 1.8 to 11 ng/L. TPCB concentrations in East River surface water from the mouth transect locations under flood tide conditions were similar to

ER001. Summary statistics for concentrations measured in the East River sampling program are included in Table 5-22.

Figure 5-34 shows the temporal pattern of the TPCB concentration of samples from the East River sampling program. TPCB concentrations are similar in samples from the East River and the mouth transect locations. There is no discernable relationship between TPCB concentration and depth, indicating that during flood tide conditions, the water column is well mixed. Although there is some data variability (such as one elevated concentration in September and the anomalous increase in concentrations during the January sampling event), in general, TPCB concentrations appear to vary seasonally (as discussed for the data observed in the lower portion of the Study Area in Section 4.7), with higher concentrations in the warmer months of June through August and an arithmetic average of 7.7 ng/L (ranging from 4.3 to 14 ng/L), compared to the colder months of November through February with an arithmetic average of 3.6 ng/L (ranging from 0.24 to 8.8 ng/L).

Estimated particulate phase TPCB concentrations in the East River from ER001 had an arithmetic average and median of 0.15 mg/kg, with a range of 0.038 to 0.23 mg/kg. TPCB concentrations in surface water from the mouth transect locations were similar to ER001. Summary statistics for concentrations measured in the East River sampling program are included in Table 5-23.

Figure 5-35 is a box plot of the estimated particulate phase surface water concentrations in samples from the East River program, the Study Area, and Phase 2 reference areas during dry weather. Particulate phase PCB concentrations are slightly less in samples from the East River than in samples from the Study Area. Concentrations in the East River are greater than in three of the four Phase 2 reference areas and have similar particulate phase concentrations to the samples from Westchester Creek.

5.3.5 Cu

Cu concentrations in surface water under flood tide conditions from ER001 have an arithmetic average of 5.0 µg/L, a median of 4.8 µg/L, and a range of non-detect (several results at 2.0 µg/L) to 9.0 µg/L. Cu concentrations in surface water from the mouth transect locations under flood tide conditions were similar to ER001. Note that 38% of the samples from the East River program were non-detect for Cu. Summary statistics for concentrations measured in the East River sampling program are included in Table 5-24. Detected concentrations of Cu in the East River program averaged 1.9 times the MDL, so an increased frequency of results less than the detection limit is not unexpected. Appendix Bi includes the data usability assessment, which determined that all the Cu data are usable.

Figure 5-36 shows the temporal pattern of the Cu concentration of samples from the East River sampling program. Concentrations of Cu in samples from the East River and the mouth transect location are similar. There is no discernable relationship between Cu concentration and depth, indicating that under flood tide conditions, the water column is well mixed. Unlike TPAH (17) and TPCB, Cu concentrations do not appear to change seasonally, and month-to-month arithmetic averages are within a factor of 2 of one another.

Calculated particulate phase Cu concentrations in the East River from ER001 had an arithmetic average of 110 mg/kg, a median of 76 mg/kg, and a range of 24 to 330 mg/kg.[189] Cu concentrations in surface water from the mouth transect locations under flood tide conditions were similar to ER001. Summary statistics for particulate phase concentrations calculated for the East River and the transect location samples are included in Table 5-25.

Figure 5-37 is a box plot of the calculated particulate phase surface water concentrations for Cu in samples from the East River program, the Study Area, and Phase 2 reference areas during dry weather. Particulate phase Cu concentrations from the East River are similar to the downstream reaches of Newtown Creek (CM 0 – 1 and CM 1 – 2) and the downstream tributaries. Cu concentrations from the East River are less than the most upstream reach of Newtown Creek (CM 2+) and Maspeth Creek. Cu concentrations in the East River are also similar to samples collected in the four Phase 2 reference areas.

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[189] The relatively larger fraction of non-detects for Cu for both whole-water (total) and filtered (dissolved) samples imparts additional variability in the calculated particulate phase concentrations. Specifically, because the Cu concentrations are relatively low and near the detection limits, the reported dissolved concentration is greater than the associated total concentrations for several samples.

5.3.6 Summary of East River Data

OC, TSS, TPAH (17), TPCB, and Cu concentrations measured in the East River water column under flood tide conditions (and at the mouth of Newtown Creek) do not show differences with respect to depth. This suggests that the water column is generally vertically well mixed in this area. TPAH (17) and TPCB appear to vary seasonally, with higher concentrations in the warmer months relative to colder months. TOC, TSS, and Cu do not appear to vary seasonally. Concentrations of TPAH (17), TPCB, and Cu in the East River are similar to the concentrations measured in the lower 2 miles of the Study Area, suggesting that this portion of Newtown Creek is strongly affected by East River tidal water.

5.4 Shoreline Erosion

Contaminated shoreline soils can release contaminants directly to the Study Area through shoreline erosion. Susceptibility to shoreline erosion increases when erodible soils are exposed to surface water currents, stormwater runoff, waves, vessel wakes, shoreline seeps, wind, and over-steepened bank conditions. Shoreline armoring or the presence of stabilizing vegetation can reduce the potential for shoreline erosion, but if shoreline structures are in disrepair or vegetation is distressed, migration of impacted soil into the creek can occur.

The evaluation of the shoreline erosion pathway was originally conducted during the other non-point sources evaluation described in the Phase 2 RI Work Plan Volume 2 (Anchor QEA 2014b) and included the evaluation of shorelines both within the Study Area (below the OHW elevation of 1.81 feet in NAVD88) and outside the Study Area (above the OHW). This initial evaluation was based on review of available documents, including Sanborn maps and spill records, and observations of bank conditions, including identification of undermined banks, steepness, failing structures or objects (e.g., pavement, fences, or bulkheads), and extent of vegetation—all relative to exposed soils. Observations of individual upland sites were conducted during wet and dry weather conditions and (in many cases) included observations during multiple seasons and tidal elevations.

Upon review of the RI data, USEPA determined that potentially erodible shorelines needed further investigation and characterization to support the FS (Sivak 2016). The goal of the shoreline sediment sampling program conducted during Part 1 of the FS field program in

2017 was to further characterize contaminant distributions in (or near) potentially erodible shorelines for development and screening of remedial alternatives in the FS. Potentially erodible shorelines within the Study Area are shown in Figure 5-38. The FS Part 1 shoreline sampling program directly targeted shorelines below the OHW with the potential to erode (defined as exposed soils on the shoreline) that had elevated sediment concentrations nearby and a lack of preexisting shoreline or nearshore data collected during the RI. The Newtown Creek shoreline is defined as the intertidal area around the perimeter of Newtown Creek and its tributaries from the MLLW elevation (-2.61 feet in NAVD88) to the OHW. Depending on the slope, the shoreline width varies from approximately 10 to 40 feet. The shoreline is a relatively small component of the Study Area, representing approximately 5% of its total surface area, as shown in Figure 5-39. Material within the shoreline area likely represents a mix of native soils, fill associated with reworking of shorelines by the adjacent upland site owners or occupants, material deposited from multiple other sources within Newtown Creek (e.g., point source discharges and shoreline seeps) or outside of Newtown Creek (e.g., solids from the East River), and various other fill materials, including materials likely derived from outside the Study Area.

Sampling was conducted to meet the objectives of the field program outlined in Section 3 of the FS FP Work Plan (Anchor QEA 2017a), using the methods described in Section 5.1 of the FS FSAP (Anchor QEA 2017b). These data were collected to supplement RI shoreline sediment data that had already been collected as part of the Phase 1 and 2 programs.[190] A total of 94 RI sediment samples were collected in or near shorelines[191] during the RI and Part 1 of the FS field programs to characterize shoreline sediment. These shoreline data are the focus of this section. Because all shoreline data were collected from below the OHW elevation and are within the Study Area, the data presented in this section represent only the shoreline conditions within the Study Area.

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[190] As discussed in Section 2, data from Part 1 of the FS field program are included as part of the RI dataset. As such, any general references in this report to the RI dataset implicitly include these FS data, unless otherwise specified.

[191] Sediment samples considered in or near shorelines include RI surface sediment data within the potentially erodible shorelines, RI surface sediment data within 30 feet of the OHW of potentially erodible shorelines, and FS shoreline sediment data.

5.4.1 Shoreline Sampling Locations

Sediment samples collected in or near shoreline areas as part of the RI and Part 1 of the FS field activities are shown in Figure 5-39. During the FS Part 1 field program, 35 shoreline sediment and in-water surface sediment samples near shorelines were collected, representing 23 shoreline locations.[192] During Phase 1 and Phase 2 field activities, a total of 59 surface sediment samples were collected from the shoreline areas throughout the Study Area to delineate the nature and extent of contaminants in the Study Area and support the ecological and human health risk assessments.

5.4.2 Shoreline Surface Sediment Sampling Results

The shoreline surface sediment results for TPAH (17), TPCB, and Cu collected during the RI and Part 1 of the FS field activities are presented in Sections 5.4.2.1 through 5.4.2.3. Sediment concentrations for TPAH (17), TPCB, and Cu, measured in shoreline sediment samples in or near potentially erodible shorelines are presented in Figures 5-40 through 5-42 and in Tables 5-26 through 5-28.

5.4.2.1 Surface TPAH (17)

TPAH (17) concentrations in surface sediment in or near potentially erodible shorelines in each reach of the Study Area are summarized in Table 5-26. In addition, a longitudinal profile comparing surface sediment concentrations in or near potentially erodible shorelines to other nearby RI surface sediment data is provided in Figure 5-40.

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[192] The focus of the shoreline sediment sampling effort during the FS Part 1 field program was to collect surface sediment samples, but subsurface sediment samples on potentially erodible shorelines were also collected in some instances. Shoreline sediment sampling was performed where feasible at two intervals: surface (0 to 15 cm [0 to 6 inches]) and subsurface (15 to 60 cm [6 to 24 inches]). In instances where a shoreline sediment sample could not be collected, a surface sediment sample (0 to 15 cm [0 to 6 inches]) was collected near the shoreline below the MLLW, and a subsurface sample was not collected.

The range, arithmetic average, and median for each reach and any notable patterns in the shoreline sediment data (as compared to other nearby sediment data) are as follows (if patterns are not discussed, it means no notable patterns were observed)[193]:

- **CM 0 – 1**. Only one shoreline sediment sample was collected in the first mile of the Study Area, so statistics are not provided. The TPAH (17) concentration of that sample (14 mg/kg) is within the range of the surface sediment data in CM 0 – 1 (range of 2.2 to 570 mg/kg).
- **CM 1 – 2**. TPAH (17) concentrations range from 0.58 to 36 mg/kg (arithmetic average = 24 mg/kg; median = 26 mg/kg) and are consistent with other surface sediment data in CM 1 – 2 (range of 13 to 94 mg/kg), with one exception. One sample collected from the shoreline adjacent to LIRR – 47th Avenue (0.58 mg/kg) is lower than the other surface sediment data in CM 1 – 2.
- **CM 2+**. TPAH (17) concentrations range from 0.24 to 1,400 mg/kg (arithmetic average = 120 mg/kg; median = 18 mg/kg). Concentrations are consistent with, or lower than, other surface sediment data in CM 2+, which range from 25 to 1,200 mg/kg; with the exception of the maximum TPAH (17) surface sediment concentration within the Study Area (1,400 mg/kg); this was measured in a sample collected from the shoreline sediments adjacent to Cipico Construction, Inc.
- **English Kills**. TPAH (17) concentrations range from 6.5 to 190 mg/kg (arithmetic average = 70 mg/kg; median = 61 mg/kg). Concentrations fall within the range of other surface sediment data in English Kills (range of 5.9 to 290 mg/kg), but localized elevated TPAH (17) concentrations exist in a number of samples collected in or near the shoreline. The highest localized TPAH (17) surface sediment concentrations in or near the shoreline within English Kills are measured in samples collected from sediments adjacent to B.C.F. Oil (DAR No. 27) (190 mg/kg), from sediments adjacent to Darrel Albers Property (130 mg/kg), and from within the sediment mound outside the NCB-015 outfall at the head of English Kills adjacent to LIRR – Johnson Avenue (130 mg/kg).

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[193] The profile shown in Figure 5-40 is presented to provide comparisons between shoreline sediment data and other sediment data and is not intended to assess potential longitudinal patterns in shoreline TPAH (17) concentrations. Overall patterns of surface sediment data (including shoreline sediment data) are presented in Section 4.2.3.

- **East Branch**. TPAH (17) concentrations range from 2.4 to 98 mg/kg (arithmetic average = 39 mg/kg; median = 35 mg/kg) and fall within the range of, or lower than, other surface sediment data in East Branch, which range from 6.3 to 370 mg/kg.
- **Maspeth Creek**. TPAH (17) concentrations range from 5.6 to 180 mg/kg (arithmetic average = 63 mg/kg; median = 50 mg/kg) and are generally consistent with other surface sediment data in Maspeth Creek (range of 49 to 71 mg/kg), with the exception of the highest concentrations measured in Maspeth Creek in surface sediment samples. These are located along the southern shoreline of Maspeth Creek adjacent to Roadway Package Systems, Inc., and Scott Realty Co. (150 and 180 mg/kg, respectively).
- **Dutch Kills**. TPAH (17) concentrations range from 0.87 to 120 mg/kg (arithmetic average = 49 mg/kg; median = 55 mg/kg) and fall within the range of other surface sediment data in Dutch Kills (range of 12 to 130 mg/kg), with the exception of the samples collected from the shorelines of AA Truck Renting Corp (0.87 mg/kg) and LD Realty Corp (11 mg/kg), which have concentrations lower than the nearby surface sediment data.

In summary, shoreline surface sediment TPAH (17) concentrations are generally similar to, or lower than, the rest of RI surface sediments collected in the same areas. However, a sample collected from the shoreline sediments adjacent to Cipico Construction, Inc., in CM 2+, contains the highest TPAH (17) concentration measured in surface sediments in the Study Area; other localized elevated shoreline surface sediment TPAH (17) concentrations exist in Dutch Kills, Maspeth Creek, and English Kills.

5.4.2.2 Surface TPCB

TPCB concentrations in surface sediment in or near potentially erodible shorelines in each reach of the Study Area are summarized in Table 5-27. In addition, a longitudinal profile comparing surface sediment concentrations in or near potentially erodible shorelines to other RI surface sediment data is provided in Figure 5-41.

The range and arithmetic average for each reach and any notable patterns in the shoreline sediment data (as compared to other nearby sediment data) are as follows (if patterns are not discussed, it means no notable patterns were observed)[194]:

- **CM 0 – 1**. Only one shoreline sediment sample was collected in the first mile of the Study Area, so statistics are not provided. However, the TPCB concentration of that sample (0.65 mg/kg) is within the range of the surface sediment data in CM 0 – 1 (range of 0.12 to 3.0 mg/kg).
- **CM 1 – 2**. TPCB concentrations range from 0.098 to 1.8 mg/kg (arithmetic average and median = 0.94 mg/kg) and generally fall within the range of other surface sediment data in CM 1 – 2 (range of 0.49 to 3.0 mg/kg).
- **CM 2+**. Shoreline sediment TPCB concentrations range from 0.019 to 160 mg/kg (arithmetic average = 13 mg/kg; median = 0.79 mg/kg). Most concentrations are lower than other surface sediment data in CM 2+, which range from 1.0 to 90 mg/kg, with the exception of the maximum concentrations observed in surface sediment in the main stem. The second highest TPCB concentration measured in a shoreline sample in the Study Area was collected adjacent to Cipico Construction, Inc (160 mg/kg).
- **English Kills**. TPCB concentrations range from 0.11 to 38 mg/kg (arithmetic average = 12 mg/kg; median = 9.4 mg/kg). Concentrations fall within the range of, or lower than, other surface sediment data in English Kills, which range from 0.59 to 63 mg/kg, but localized elevated TPCB concentrations exist in a number of samples collected in or near the shoreline within English Kills. The highest TPCB surface sediment concentrations within English Kills were measured in shoreline samples collected from within the sediment mound outside the NCB-015 outfall at the head of English Kills, adjacent to LIRR – Johnson Avenue (38 mg/kg), and from sediments adjacent to Darrel Albers Property (32 mg/kg).
- **East Branch**. TPCB concentrations range from 0.033 to 16 mg/kg (arithmetic average = 2.8 mg/kg; median = 0.99 mg/kg) and generally fall within the range of other surface sediment data in East Branch (range of 0.024 to 12 mg/kg).

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[194] The profile shown in Figure 5-41 is presented to provide comparisons between shoreline sediment data and other sediment data and is not intended to assess potential longitudinal patterns in shoreline TPCB concentrations. Overall patterns of surface sediment data (including shoreline sediment data) are presented in Section 4.2.3.

- **Maspeth Creek**. TPCB concentrations range from 0.29 to 25 mg/kg (arithmetic average = 2.7 mg/kg; median = 1.1 mg/kg). Concentrations are consistent with, or lower than, other surface sediment data in Maspeth Creek, which range from 0.95 to 15 mg/kg, with the exception of the highest concentration measured in Maspeth Creek in a surface sediment sample located along the southern shoreline (25 mg/kg).
- **Dutch Kills**. TPCB concentrations range from 0.015 to 380 mg/kg (arithmetic average = 30 mg/kg; median = 3.7 mg/kg). Elevated TPCB concentrations, including the highest concentration measured in Study Area surface sediment, exist in samples taken in the vicinity of Dutch Kills CM 1.2, including samples collected from shoreline sediment adjacent to Borden Realty Co. The maximum concentration in Dutch Kills (380 mg/kg) is two orders of magnitude higher than the maximum concentration in other surface sediment in Dutch Kills (8.4 mg/kg).

Shoreline surface sediment TPCB concentrations are generally similar to, or lower than, the rest of the RI surface sediments collected in the same general areas. However, samples collected from the shoreline sediments adjacent to Borden Realty Co. in Dutch Kills and Cipico Construction, Inc., in CM 2+ contain the highest TPCB concentrations measured in surface sediments; other localized elevated TPCB concentrations exist in English Kills and Maspeth Creek.

5.4.2.3 Surface Cu

Cu concentrations in surface sediment in or near potentially erodible shorelines in each reach of the Study Area are summarized in Table 5-28. In addition, a longitudinal profile comparing surface sediment concentrations in or near potentially erodible shorelines to Phase 1 and Phase 2 surface sediment data is provided in Figure 5-42.

The range and arithmetic average for each reach and any notable patterns in the shoreline sediment data (as compared to other nearby sediment data) are as follows (if patterns are not discussed, it means no notable patterns were observed)[195]:

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[195] The profile shown in Figure 5-42 is presented to provide comparisons between shoreline sediment data and other sediment data and is not intended to assess potential longitudinal patterns in shoreline Cu concentrations. Overall patterns of surface sediment data (including shoreline sediment data) are presented in Section 4.2.3.

- **CM 0 – 1**. Only one shoreline sediment sample was collected from the first mile of the Study Area, so statistics are not provided. The Cu concentration of that sample (96 mg/kg) is within the range of the surface sediment data in CM 0 – 1 (range of 11 to 780 mg/kg).
- **CM 1 – 2**. Cu concentrations range from 60 to 320 mg/kg (arithmetic average = 230 mg/kg; median = 260 mg/kg) and generally fall within the range of other surface sediment data in CM 1 – 2 (range of 69 to 650 mg/kg).
- **CM 2+**. Cu concentrations range from 39 to 2,800 mg/kg (arithmetic average = 720 mg/kg; median = 410 mg/kg) and are on the lower end of the range of, or lower than, other surface sediment data in CM 2+, which range from 120 to 37,000 mg/kg.
- **English Kills**. Cu concentrations range from 60 to 2,400 mg/kg (arithmetic average = 630 mg/kg; median = 530 mg/kg) and are consistent with, or lower than, other surface sediment data in English Kills, which range from 150 to 4,300 mg/kg.
- **East Branch**. Cu concentrations range from 42 to 6,300 mg/kg (arithmetic average = 890 mg/kg; median = 250 mg/kg) and are generally consistent with, or higher than, other surface sediment data in East Branch, which range from 32 to 1,400 mg/kg. However, shoreline sediment sample results are both higher and lower than nearby surface sediment data in localized areas. The maximum Cu surface sediment concentration (6,300 mg/kg) in or near a potentially erodible shoreline was measured in a sample collected in East Branch around CM 3.1, near Feldman Metropolitan (DAR No. 217).
- **Maspeth Creek**. Cu concentrations in shoreline sediment range from 72 to 5,900 mg/kg (arithmetic average = 780 mg/kg; median = 340 mg/kg). Most of the Cu concentrations in shoreline samples are within the range of other surface sediment in Maspeth Creek (400 to 730 mg/kg), although there are a few individual results that are either higher or lower than this range. The arithmetic average of the shoreline sediment data in Maspeth Creek is higher than the maximum concentration result of other surface sediment data within Maspeth Creek. The two highest Cu concentrations in or near a potentially erodible shoreline within Maspeth Creek were measured in shoreline sediment samples located along the southern shoreline adjacent to Roadway Package Systems, Inc., and Scott Realty Co. (1,600 and 5,900 mg/kg, respectively).
- **Dutch Kills**. Cu concentrations range from 13 to 850 mg/kg (arithmetic average and median = 370 mg/kg). Most of the Cu concentrations in shoreline samples are within the

range of other surface sediment in Dutch Kills, which range from 110 to 730 mg/kg, although there are a few individual results that are either higher or lower than this range.

Most of the shoreline surface sediment Cu concentrations are generally within the range of the rest of the RI surface sediments collected in the same general areas. Localized elevated Cu concentrations in shoreline surface sediments are observed in areas of East Branch and Maspeth Creek. Localized Cu concentrations in shoreline surface sediments that are lower than other nearby surface sediment are observed throughout the tributaries and CM 2+.

5.4.3 Shoreline Subsurface Sediment Sampling Results

The focus of the shoreline sediment sampling effort during the FS Part 1 field program was to collect surface sediment samples, but subsurface sediment samples on potentially erodible shorelines were also collected at locations where shoreline surface sediment samples above the MLLW were collected. One subsurface sediment sample was collected at the 15- to 60-cm (6- to 24-inch) interval at 24 locations. The shoreline subsurface sediment results for TPAH (17), TPCB, and Cu collected during the FS Part 1 field activities are presented in Sections 5.4.3.1 through 5.4.3.3. Sediment concentrations for TPAH (17), TPCB, and Cu measured in shoreline surface and subsurface sediment samples are presented in Figures 5-43 through 5-45.

5.4.3.1 Subsurface TPAH (17)

Core profiles comparing shoreline surface sediment and subsurface sediment TPAH (17) concentrations are provided in Figure 5-43. In general, TPAH (17) subsurface concentrations are approximately within a factor of 2 to 3 of surface concentrations (13 of the 24 cores have higher surface concentrations than subsurface; the reverse is true for the other 11 cores). TPAH (17) concentrations in both surface and subsurface samples generally are similar to or less than other samples collected within the same general area. However, relatively high TPAH (17) concentrations (when compared to other surface sediment samples collected throughout the Study Area) exist in subsurface samples collected from shorelines within English Kills adjacent to LIRR – Johnson Avenue (EK135, which is within the sediment mound outside the NCB-015 outfall) and at Frito Lay II (EK133); within Dutch Kills at Borden Realty Co. (DK066); and within CM 2+ at Cipico Construction, Inc. (NC386 and NC387) and Pebble Lane Associates (NC388). One of the two samples collected from the shoreline sediments adjacent to Cipico Construction, Inc. (NC386), in CM 2+, contains the highest TPAH (17) concentration measured in both surface and subsurface shoreline sediments (1,400 mg/kg and 2,140 mg/kg, respectively) collected during the FS Part 1 field program.

5.4.3.2 Subsurface TPCB

Core profiles comparing shoreline surface sediment and subsurface sediment TPCB concentrations are provided in Figure 5-44. Similar to TPAH (17), in general, shoreline subsurface sediment TPCB concentrations are within a factor of 2 to 3 of surface concentrations (14 of the 24 cores have higher surface concentrations than subsurface; the reverse is true for the other 10 cores). TPCB concentrations in both surface and subsurface samples generally are similar to or less than other samples collected within the same general area. However, relatively high TPCB concentrations (when compared to other surface sediment samples collected throughout the Study Area) exist in subsurface samples collected in the shorelines within English Kills adjacent to LIRR – Johnson Avenue (EK135, which is within the sediment mound outside the NCB-015 CSO outfall) and adjacent to Frito Lay II (EK133) and Darrell Albers Property (EK131); within Dutch Kills adjacent to Borden Realty Co. (DK066); and within CM 2+ adjacent to LIRR – 47th Avenue (NC380) and Cipico Construction, Inc. (NC386 and NC387, respectively). EK135 has the highest TPCB subsurface sediment concentration (182 mg/kg) collected in subsurface shoreline sediment during the FS Part 1 field program.

5.4.3.3 Subsurface Cu

Core profiles comparing shoreline surface sediment and subsurface sediment Cu concentrations are provided in Figure 5-45. Similar to TPAH (17) and TPCB, in general, shoreline subsurface sediment Cu concentrations are within a factor of 2 to 3 of surface concentrations (7 of the 24 cores have higher surface concentrations than subsurface; the reverse is true for the other 17 cores). Cu concentrations in both surface and subsurface samples generally are similar to (or less than) other samples collected within the same general area. However, relatively high Cu concentrations exist in subsurface samples collected in the shorelines within Dutch Kills adjacent to Borden Realty Co. (DK066), within English Kills adjacent to LIRR – Johnson Avenue (EK135, which is within the sediment mound outside the NCB-015 CSO outfall) and adjacent to Frito Lay II (EK133); and within

CM 2+ adjacent to Cipico Construction, Inc. (NC386). The shoreline sample (EK135) collected within the sediment mound outside the NCB-015 CSO outfall adjacent to LIRR – Johnson Avenue has the highest Cu subsurface sediment concentration (8,870 mg/kg) collected in surface and subsurface shoreline sediment during the FS Part 1 field program.

5.4.4 Shoreline Erosion Summary

Because all shoreline data were collected from below the OHW elevation and are within the Study Area, the data summary presented in this section represents only the shoreline conditions within the Study Area.

With a few exceptions, TPAH (17), TPCB, and Cu concentrations from surface sediment samples in or near potentially erodible shorelines are within the range of, or lower than, the rest of the RI surface sediments collected in the same general area. This indicates that, in general, shoreline erosion is not a significant ongoing source of contamination. Surface sediment samples collected near potentially erodible shorelines show a similar trend to sediment samples collected during the RI, with a generally increasing trend in concentration in the upstream direction. The exceptions to this general trend include the shorelines adjacent to Cipico Construction, Inc., and Borden Realty Co., where surface sediment concentrations of TPAH (17) and TPCB, respectively, are higher than any other surface sediment samples collected as part of the RI. Elevated concentrations in (or near) potentially erodible shorelines were also measured in surface sediments adjacent to B.C.F. Oil (DAR No. 27); LIRR – Johnson Avenue; Darrel Albers Property; Feldman Metropolitan; Roadway Package Systems, Inc.; and Scott Realty Co. Sediments from these sites where elevated concentrations in (or near) potentially erodible shorelines were identified have the potential to erode and migrate within the creek.

At most locations where surface and subsurface sediment concentrations were collected at shoreline sampling locations, concentrations for TPCB, TPAH (17), and Cu are relatively similar (within a factor of 2 or 3) in surface and subsurface sediment. There are some locations, however, where subsurface concentrations are more than three times greater than surface concentrations at the same location. Specifically, shoreline samples collected within the sediment mound outside the NCB-015 outfall adjacent to the LIRR-Johnson Avenue property

(EK135) have the highest TPCB and Cu subsurface shoreline sediment concentrations collected during the FS Part 1 field program. This is also the case in subsurface shoreline sediments near Cipico Construction, Inc. (NC386), which have the highest TPAH (17) subsurface shoreline sediment concentrations collected during the FS Part 1 field program. Elevated TPAH (17) and TPCB concentrations in subsurface sediments were also measured in shoreline sediments adjacent to Frito Lay II (EK133), Borden Realty Co. (DK066), Cipico Construction, Inc. (NC386 and NC387), Pebble Lane Associates (NC388), Darrell Albers Property (EK131), and LIRR – 47th Avenue.

With the few exceptions noted previously, concentrations of TPCB, TPAH (17), and Cu in shoreline sediment samples are, in general, within the range of, or lower than the RI surface sediments collected in the same general area. This indicates that, in general, shoreline erosion below the OHW is not a significant ongoing source of contamination to the Study Area. Elevated TPAH (17) and TPCB shoreline surface sediment concentrations adjacent to Cipico Construction, Inc., and Borden Realty Co., as well as elevated PCB and Cu shoreline subsurface sediment concentrations collected within the sediment mound outside the NCB-015 outfall adjacent to the LIRR-Johnson Avenue property (EK135) suggest potential upland sources of contamination to the Study Area. However, because sample collection was limited to within the Study Area (below OHW), it is not possible to determine whether the upland sites are a source to the Study Area. These properties were referred to USEPA for consideration of additional uplands investigations, outside the scope of the RI/FS, to ascertain whether there are ongoing sources from these properties that would require control prior to remedial action.

5.5 Atmospheric Deposition

Annual chemical loads to the Study Area due to atmospheric deposition were calculated using regional atmospheric deposition fluxes from data collected at stations throughout New Jersey and reported in 2004 by the New Jersey Atmospheric Deposition Network (NJADN), on behalf of the New Jersey Department of Environmental Protection (Reinfelder et al. 2004). This approach is consistent with atmospheric loads incorporated into the CARP Contaminant Fate and Transport and Bioaccumulation Sub-models (CARP 2007) that were developed for the NY/NJ Harbor urban estuary. To calculate annual atmospheric deposition loads in the Study Area, the atmospheric flux data from the NJADN station located at Liberty State Park in Jersey City, New Jersey, were used. This station was also used to represent loads to the East River in the CARP model. The NJADN flux data included gas absorption, dry particle deposition, and precipitation deposition. Gas absorption is a process that is not observed in metals, so it was not included for Cu.

A total annual atmospheric deposition flux was calculated by summing the three terms (i.e., gas absorption [excluding Cu], dry particle deposition, and precipitation deposition). The annual atmospheric deposition load was then calculated by multiplying the total annual atmospheric flux for a given chemical by the plan view surface area for each reach of the Study Area. In the NJADN data, total PAH (indicated here as TPAH [36]) included 36 PAH chemical constituents, 14 of which were included in the TPAH (17) sum used in Newtown Creek. The calculated TPAH (36) atmospheric loads were not adjusted, because it is likely that the NJADN estimate is an overestimate of TPAH (17) loads, so it is more conservative for the purposes of this evaluation.

The fluxes for Jersey City reported by Reinfelder et al. (2004) are presented in Table 5-29, along with the atmospheric deposition loads estimated for Newtown Creek by reach and summed over the entire Study Area, for TPAH (36), TPCB, and Cu.

Using point sources[196] and groundwater loads[197] as a basis for comparison, the estimated annual atmospheric deposition loads for the three chemicals in Newtown Creek represent a relatively small ongoing source of contamination to the Study Area (see Table 5-29). The Study Area-wide annual atmospheric load of TPAH (36) was estimated to be 5.0 kg/year, which is 10 to 11 times lower than the point source load (52 to 55 kg/year; see Section 5.1.4.3), and 170 to 300 times lower than the groundwater load (830 to 1,500 kg/year; see Section 5.2.2). The annual atmospheric load of TPCB was estimated to be 0.041 kg/year, which is 11 to 21 times lower than the point source load (0.43 to 0.84 kg/year), and similar to, or seven times lower than, the groundwater load (0.045 to 0.27 kg/year). The annual atmospheric load of Cu was estimated to be 3.4 kg/year, which is 120 to 140 times lower than

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[196] As discussed in Section 5.1.4.3.5, the results from the RUM/SOM hybrid and CDM were used to establish ranges of point source load estimates; these ranges were used in this section for comparison to atmospheric deposition loads (see Table 5-29).

[197] As discussed in Section 5.2.2, a range of estimated groundwater loads was developed for TPAH (17) and TPCB.

the point source load (390 to 460 kg/year) and 3 times lower than the groundwater load (10 kg/year). Atmospheric loads were also calculated on a reach-specific basis (see Table 5-29) to allow them to be compared with other loads in the mass balance evaluation presented in Section 6.5.

5.6 Overwater Activities

The evaluation of overwater activities within the Study Area was conducted during the other non-point sources evaluation, as part of the draft SSAM (Anchor QEA 2014m). Site operations that occur over water—such as loading/unloading, maintenance and repairs at creek-side docks, wharves, and piers—represent potential sources of chemicals to the Study Area. Discharges from vessels (e.g., gray water, bilge water, and ballast water), fuel releases, and spills during loading or unloading of bulk dry or liquid cargo operations are also considered overwater activities. Temporary construction and repair activities were considered minor and were not included in the evaluation.

The activities most commonly associated with overwater uses in the Study Area are petroleum handling and scrap metal hauling. The nine sites identified as having current overwater activities fall into one of these two categories. The petroleum handling sites include Bayside Fuel Oil Depot (DAR No. 51), BP Products N America Brooklyn Terminal (DAR No. 48),[198] Motiva Brooklyn Terminal (DAR No. 50), and Metro Terminal (DAR No. 52), and the scrap metal hauling sites include Masluf Realty Corp. (DAR No. 201), Allocco Recycling Corp. (DAR No. 203), Charles J. King, Inc. (DAR No. 206), Hugo Neu (DAR No. 125), and T&T Scrap, LLC. There are also recreational boats docked along the Queens side of Newtown Creek, west of the Pulaski Bridge, but it is presumed that these boats are not associated with a particular site. It is also possible that sporadic, unintentional overwater sources may occur at other sites, as well, but these instances are considered to be infrequent and consequently of little significance. Descriptions of the overwater activities at these sites have been documented in Table E4-1 of the draft SSAM (Anchor QEA 2014m).

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[198] BP Products N America Brooklyn Terminal was sold to Kinder Morgan Operating L.P. "D", a Delaware limited partnership effective February 2, 2016. Nonetheless, to maintain consistency with the draft DAR, this facility is referred to as "BP Products N America Brooklyn Terminal."

Overwater spills are unpermitted releases that occur directly into the creek and can be intentional or unintentional. As described in Section 3.2.11, the NYSDEC Spill Incidents Database contains information for reported spills to surface water and details regarding the cause and source of a spill and subsequent investigations, if any. The information available in the database is variable. It can be difficult to decipher where a release occurred, and the source or cause of a spill may be unknown. When the spills to surface water are linked to a shoreline site, they have been documented in Table E4-1 of the draft SSAM (Anchor QEA 2014m).

As of December 2008, incidental discharges (e.g., gray water, bilge water, and ballast water) from vessels greater than 79 feet in length are required to be permitted under USEPA's Small Vessel General Permit (sVGP). The sVGP applies to discharges incidental to the normal operations of non-recreational vessels and contains effluent limits for different types of discharges. There are additional effluent limits pertaining to spill containment and cleanup associated with barges (e.g., fuel and dry bulk cargo). Non-military, non-recreational vessels less than 79 feet in length became regulated under the sVGP in 2014. With these permits in place, it is expected that incidental discharges from the barges making deliveries or loading at the sites in the Study Area have been, and will be, further reduced.

Contaminant releases from overwater activities (e.g., material transfer, maintenance, repair, and operations at docks, wharves, or piers), discharges from vessels, fuel releases, and spills are expected to have been more significant historically, prior to the implementation of improved BMPs. A discussion of historical spills is presented in Section 3.2.11. Regulations and BMPs are likely to have reduced such contributions of contamination to surface water or sediment over time. Many facilities are now required to maintain spill prevention plans and have instituted practices to reduce spills while loading and unloading vessels. Although improved BMPs and spill plans are likely to have reduced the occurrence of overwater releases significantly, it is acknowledged that current and future releases could occur. Based on the minimal record of releases associated with overwater activities and the regulations and BMPs in place, the relative magnitude of releases from overwater activities compared to other source pathways is believed to be minor. Additionally, overwater releases are not necessarily measurable. Thus, overwater activities are not considered a significant ongoing source to the Study Area, although the potential for such activities to result in a future significant release exists, due to the nature of the activities.

5.7 Shoreline Seeps

Intertidal shoreline seeps (fluid emerging from the shoreline) and bulkhead seeps (fluid observed flowing out from bulkheads around joints, bolts, cracks, or holes) are a potential ongoing source to the Study Area, particularly if indications of contamination (such as visual signs of sheen, NAPL, or discoloration of surface water) are associated with the shoreline or bulkhead seep.

No observable contaminant seeps were documented during the Phase 1 and Phase 2 field activities. Bulkhead seeps were observed during Phase 1 dry weather surveys (Anchor QEA 2012o) and other RI field work. This flow was typically observed within the intertidal zone and may have been creek water that had inundated fill materials, gaps, or voids behind bulkheads during high tide and then drained via gravity when the tide receded; this is referred to as bank storage. Visual indications of contamination, such as sheen, were not observed at these locations.

During the 2016 gas ebullition field survey, a sheen seep was observed originating from the Waste Management of NY/Steel Equities (formerly Pratt Oil Works; DAR No. 56) bulkhead. In 2019, with the property owner's approval, ExxonMobil was responsible for the installation of a sheetpile wall along the bank in the area of the reported seep observation, which has eliminated the potential for seeps in this area (see Section 4.1.2.1 of Appendix D). The property owner is responsible for future maintenance of the wall.

During FS Part 1 field activities, a visual survey was performed during low tide along the shoreline at each shoreline sediment sampling location (described in Section 5.4). If a shoreline seep was observed, an opportunistic seep sample was collected (providing that access allowed for safe sampling). Observed seeps were located between elevation 0.8 and -1.4 feet in NAVD88 and were often only visible within 1 or 2 hours of low tide. Seeps were sampled consistent with Section 3 of the FS FP Work Plan (Anchor QEA 2017a), using the methods described in Section 5.1 of the FS FSAP (Anchor QEA 2017b). Opportunistic seep sample locations are shown in Figure 5-46 and further discussed in Sections 5.7.1 and 5.7.2, as well as in Section 6.4 of Appendix F.

Observations of seeps emanating from bulkheads and shorelines in the Study Area are discussed in Section 5.7.3.

5.7.1 Opportunistic Seep Sample Locations

During the FS Part 1 shoreline sediment sampling program, 10 shoreline seeps were identified along the following four shoreline properties within the Study Area:

- Along the Queens shoreline of Newtown Creek, downstream of the Turning Basin near CM 2.1 at LIRR – 57th Avenue (three seeps observed)
- Along the Queens shoreline of Newtown Creek, upstream of Maspeth Creek near CM 2.6 at Cipico Construction, Inc. (five seeps observed)
- Along the Greenpoint shoreline of English Kills near CM 3.3 at 8 Rewe Street (one seep observed)
- At the head of English Kills at LIRR – Johnson Avenue (one seep observed)

All observed seeps were sampled, with the exception of the seep at LIRR – Johnson Avenue, which was not sampled due to safety concerns with accessing the shoreline. The locations of these 10 shoreline seeps are shown in Figure 5-46.

Following the collection and analysis of the opportunistic seep samples, the concentrations of TPAH (17), TPCB, and Cu were compared to dry weather surface water concentrations collected during the RI (see Section 4.7.3). This comparison was conducted to evaluate the potential influence of shoreline seeps on surface water quality within the Study Area, and ultimately ecological and human health risks. Chloride concentrations measured in the shoreline seep samples were also compared to dry weather surface water concentrations, as an indicator of tidal surface water and to help distinguish between flow from seeps consisting of lateral discharge of fresher groundwater versus flow consisting of tidal water released as bank storage. These analyses are presented in Section 6.4 of Appendix F, and the results are summarized in Section 5.7.2.

5.7.2 Opportunistic Seep Sample Results

Opportunistic seep sample results for TPAH (17), TPCB, and Cu collected during the FS Part 1 sampling program are presented in Sections 5.7.2.1 through 5.7.2.3. Opportunistic seep sample concentrations compared to dry weather surface water concentrations for TPAH (17), TPCB, Cu, and chloride are presented in Figures 5-47 through 5-50. Analytical data for the opportunistic seep samples are presented in Table 5-30.

5.7.2.1 TPAH (17)

TPAH (17) concentrations in the nine opportunistic seep samples have an arithmetic average of 1.5 µg/L, a median of 0.15 µg/L, and a range of 0.014 to 12 µg/L. Figure 5-47 compares TPAH (17) data from the opportunistic seep samples to dry weather surface water collected during the Phase 1 and Phase 2 field activities. TPAH (17) concentrations from the opportunistic seep samples fall within the range of, or lower than, concentrations of dry weather surface water samples collected in the same reaches of the Study Area, which range from 0.0019 to 1.2 µg/L, except for the seep sampled on the shoreline of English Kills at CM 3.3 adjacent to 8 Rewe Street (12 µg/L; EK127OS). The TPAH (17) concentration measured from this seep is approximately an order of magnitude higher than the maximum concentration of dry weather surface water samples collected in English Kills (1.2 µg/L).

5.7.2.2 TPCB

TPCB concentrations in the nine opportunistic seep samples have an arithmetic average of 57 ng/L, a median of 5.5 ng/L, and a range of 1.2 to 390 ng/L. Figure 5-48 compares TPCB data from the opportunistic seep samples to dry weather surface water collected during Phase 1 and Phase 2 field activities. TPCB concentrations from the opportunistic seep samples fall within the range of, or lower than, concentrations of dry weather surface water collected in the same reaches of the Study Area, which range from 2.7 to 92 ng/L, except for the seep sampled on the shoreline at CM 2.6 adjacent to Cipico Construction, Inc. (390 ng/L; NC387OS). The TPCB concentration of the opportunistic seep sample from Cipico Construction, Inc., exceeds that of the highest dry weather surface water sample collected in the Turning Basin (92 ng/L).

5.7.2.3 Cu

Cu concentrations in the nine opportunistic seep samples have an arithmetic average of 7.1 µg/L, a median of 5.0 µg/L, and a range of 2.0 to 13 µg/L. Figures 5-49 compares Cu data from the opportunistic seep samples to dry weather surface water collected during Phase 1 and Phase 2 field activities. Cu concentrations from the opportunistic seep samples fall within the range of concentrations of dry weather surface water collected in the same reaches of the Study Area, which range from non-detect (1.0 µg/L) to 90 µg/L.

5.7.2.4 Chloride

Figure 5-50 compares chloride concentrations measured in the shoreline seep samples to dry weather surface water concentrations. Chloride behaves as a conservative compound (i.e., concentrations only change as a result of mixing with waters with different salinity levels and not as a result of geochemical or biological reactions) and is an indicator for surface water in a tidal system, so it is used in this evaluation to help distinguish between lateral groundwater discharge and bank storage. As shown in Figure 5-50, chloride concentrations measured in the shoreline seep samples collected from English Kills and the location downstream of the Turning Basin are similar to surface water concentrations, suggesting these shoreline seeps are more likely the result of bank storage, as opposed to lateral groundwater discharge. However, the samples collected from the Queens shoreline of the Turning Basin upstream of Maspeth Creek (adjacent to Cipico Construction, Inc.) have lower chloride concentrations than those in dry weather surface water, suggesting these samples represent a contribution from lateral groundwater discharge.

5.7.3 Seeps Observed by Others

Observations of seeps emanating from bulkheads and shorelines in the Study Areas have been reported by others.

Sheen and NAPL bulkhead seeps were reported by NYCDEP (2020) in November 2016 at the following five sites: Greenpoint Energy Center (DAR No. 32), Waste Management of NY/Steel Equities (formerly Pratt Oil Works; DAR No. 56),[199] Frito Lay (DAR No. 31), B.C.F. Oil Refining, Inc. (DAR No. 27), and Manhattan Poly Bag (DAR No. 130). These seeps reported by NYCDEP were observed during exceptionally low tides on 2 days: November 13 and 14, 2016. Seeps were typically visible only within 1 or 2 hours of low tide (NYCDEP

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[199] In 2019, with the property owner's approval, ExxonMobil was responsible for the installation of a sheetpile wall along the bank in the area of the reported seep observation at Waste Management of NY/Steel Equities (formerly Pratt Oil Works; DAR No. 56), which has eliminated the potential for seeps in this area.

2020), with lower than average low tide conditions being more conducive for seepage. Thus, the period during which NYCDEP seep observations were reported was during the most conducive period for seepage over the course of the year.[200]

The source or sources of the sheen observations reported by NYCDEP at Greenpoint Energy Center (DAR No. 32) are uncertain because it is not possible to see the shoreline bank behind the "bulkhead" and confirm the sheen origin. At the Greenpoint Energy Center, there actually is no bulkhead. The structure is a 40-foot-wide timber pile-supported loading platform, which has a timber deck covered by soil and a wood plank façade at the water's edge that only extends to just below the low water line. As such, there is about 40 feet of open water and sediment beneath the platform before the shoreline bank is encountered. It is possible that the observed sheen originated from a source or sources outside of the platform, accumulated on the surface water beneath the platform during an incoming tide, and was then released during a subsequent receding tide. It is also possible that the observed sheen originated from the sediment and/or shoreline below the platform and was released during a receding tide. Response actions to address sheen and NAPL seeps were performed under NYSDEC oversight at the other four sites where seeps were observed by NYCDEP (i.e., Waste Management of NY/Steel Equities [formerly Pratt Oil Works; DAR No. 56], Frito Lay [DAR No. 31], B.C.F. Oil Refining, Inc. [DAR No. 27], and Manhattan Poly Bag [DAR No. 130]).

NYSDEC, Newtown Creek Alliance, and others have reportedly observed sheen seeping from the bulkhead adjacent to the Morgan Oil (DAR No. 60) site, with spills to English Kills reported in 2015 and 2017 (NYSDEC Spill Nos. 1509305 and 1707515, respectively) associated with sheen (NYSDEC 2017). These seeps were not sampled as part of the RI/FS process.

In 2021, as part of an evaluation of upland properties, NYSDEC conducted a seep sampling study between August and September 2021. During that investigation, two visual seep observation

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[200] Lower low tide elevations tend to coincide with greater potential for seepage from the banks, because more of the shoreline is exposed and the hydraulic gradient from the groundwater to surface water is greatest, with the greatest potential for bank seepage typically occurring during spring tides when there is the greatest difference between high and low tide elevations. The daytime low tides on November 13 and 14, 2016 were lower than 90% of 2016 daytime low tides (31 of 414 and 15 of 414, respectively, with daytime defined as 6 a.m. to 8 p.m.). The daytime low tides on November 13 and 14, 2016 were also spring tides, with tidal swings larger than 96% of 2016 daytime tidal swings (14 of 414 and 3 of 414, respectively).

surveys were performed during spring tides, along with thermal imaging and surface water and sheen-net sampling. Sampling was performed at locations where seeps were observed during both surveys, which consisted of 21 locations. The results of the 2021 NYSDEC seep survey are provided in the *Seep Investigation Data Summary Report* (HRP 2022). In September and October 2022, NYSDEC conducted a second seep survey as part of the ongoing evaluation of upland properties. As noted previously, NYSDEC has taken response actions to address sheen and/or NAPL seeps at four upland properties where seeps have been observed.

In contrast, little evidence of sheen or NAPL seeps was observed during the Phase 1, Phase 2, and FS Part 1 field activities, as described in Section 5.7. During general on-water field activities (e.g., sediment sampling, surface water sampling, biota sampling, and travel through the Study Area), field staff maintained a visual observations log to document general visual observations of the creek and activity on and adjacent to the water, including observations of erosion/deposition and long-term controls (e.g., oil booms). Moreover, no sheen or NAPL seeps were observed during field activities occurring along shorelines where staff were explicitly looking for evidence of NAPL or sheen seeps, except for the sheen seep observed from the Waste Management of NY/Steel Equities (formerly Pratt Oil Works; DAR No. 56) bulkhead during the 2016 gas ebullition field survey at Waste Management of NY/Steel Equities (formerly Pratt Oil Works; DAR No. 56), which has since been remedied. This includes approximately 3 weeks of work during the FS Part 1 shoreline survey and sampling program, and 10 days of survey work performed during low tides as part of the gas ebullition surveys in 2015, 2016, and 2017. Bulkhead and shoreline seeps were observed during Phase 1 dry weather surveys (Anchor QEA 2012o), as well as during the FS Part 1 shoreline survey; however, sheen and NAPL seeps were not observed during these surveys. A systematic investigation of the frequency, magnitude, and chemical characteristics of seep observations reported by others has not been performed as part of the RI.[201] Therefore, the potential contribution of contaminants to the Study Area via sheen and NAPL seeps, including seeps observations reported by others, and their significance relative to remedy effectiveness will be considered during the FS.

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[201] As described previously, NYSDEC conducted Study Area-wide seep sampling studies between August and September 2021 and between September and October 2022.

5.7.4 Summary

With a few exceptions, TPAH (17), TPCB, and Cu concentrations from opportunistic seep samples measured as part of the FS Part 1 field program are generally within the range of, or lower than, concentrations of dry weather surface water samples from the Phase 1 and Phase 2 field activities. In addition, chloride concentrations in the opportunistic seep samples are in the range of surface water chloride measurements, with the exception of samples collected from the Queens shoreline of the Turning Basin upstream of Maspeth Creek (Cipico Construction, Inc.). This suggests that most of the opportunistic seep samples likely represent bank storage (i.e., water that flows into interstitial spaces on an incoming tide and seeps out of the face of the bank as the tide goes out).

There are two exceptions:

- The TPAH (17) concentration measured from the seep adjacent to 8 Rewe Street in lower English Kills (12 µg/L) is approximately an order of magnitude higher than the maximum concentration of dry weather surface water samples collected in English Kills (1.2 µg/L).
- The TPCB concentration of the opportunistic seep sample from Cipico Construction, Inc. (390 ng/L), exceeds the highest dry weather surface water sample collected in the Turning Basin (92 ng/L). In addition, the samples collected adjacent to Cipico Construction, Inc., have lower chloride concentrations than those in dry weather surface water, suggesting these samples represent a contribution from lateral groundwater discharge.

Sampling artifacts could be contributing to the high TPAH (17) and TPCB concentrations in the 8 Rewe Street and Cipico Construction, Inc., opportunistic seep samples. Elevated TPCB concentrations in the opportunistic seep sample taken at Cipico Construction, Inc., could be influenced by a locally elevated sediment TPCB concentration, either through partitioning or entrainment of solids. As shown in Figure 5-48, samples collected from Cipico Construction, Inc., contained the highest shoreline surface sediment TPCB concentrations in the Study Area. In addition, sheen was observed in the opportunistic seep sample while sampling at 8 Rewe Street, where nearby surface sediment concentrations are in the top 40% of data for TPAH

(17). These elevated sediment concentrations could be potentially contributing to the elevated concentrations measured in the seeps discharging through these sediments.

Regardless of whether the opportunistic seep samples at 8 Rewe Street and Cipico Construction, Inc., represent bank storage or lateral groundwater discharge, the elevated concentrations relative to dry weather surface water concentrations suggest the potential for a localized source to the creek at these locations. Sections 4.7.3.1 and 4.7.3.3 discuss that the highest TPAH (17) and TPCB dry weather surface water concentrations were measured in English Kills and CM 2+. The elevated concentrations of TPAH (17) and TPCB measured in seep samples from these reaches of the creek may contribute to these elevated concentrations to some extent. This contribution will be evaluated further through sensitivity analysis with the CFT model during the FS (as discussed in Section 7 of Appendix F).

6 FATE AND TRANSPORT

This section discusses the various processes that affect movement of water, sediment, and chemicals within the Study Area and how the extensive RI data collection and modeling efforts provide an understanding of these processes. The findings summarized in this section are key elements to the CSM presented in Section 8 and will provide a foundation for the updated CSM that will be included in the FS. They will also inform evaluation of remedial alternatives presented in the FS, and ultimately the selection of an effective remedy for this site by USEPA.

6.1 Introduction

To understand the fate and transport of chemicals within the Study Area, there are numerous processes that must be evaluated. These processes are shown in Figure 6-1. Along with the sources and loads of chemicals, both current (as discussed in Section 5) and historical (e.g., dating back numerous decades, as described in Section 3.2), these processes determine the distributions of chemicals in the environmental media (as described in Section 4).

Contaminants (including NAPL where present) may directly enter the Study Area surface water from the various sources discussed in Section 5. These sources include CSO outfalls, stormwater discharges, direct runoff, individually permitted point source discharges, lateral discharge of groundwater, and transport from the East River under flood tide conditions (as well as bank erosion, atmospheric deposition, overwater activities, and shoreline seeps). Hydrodynamic processes affecting CFT within the water column include upstream and downstream transport due to tidal currents, density-driven circulation, and freshwater inflow. Chemical distributions in the surface water are also affected by partitioning between the particulate and dissolved phases. Chemical reactions (e.g., degradation) can also affect chemicals observed in surface water. Additionally, chemicals in the surface water are exchanged with the surface sediment as a result of several particulate and dissolved phase processes. Particulate phase exchange processes include deposition and resuspension of sediment (driven by currents and vessel propwash). Dissolved phase exchange processes include diffusion, porewater exchange flux (including contribution from tidal effects), and porewater advection driven by groundwater flow. At locations where NAPL is present, the processes that can potentially transport NAPL from sediment to the surface water include migration via advection (driven by pressure gradients on the fluid) and gas ebullition.

Within the sediment bed, the surface sediment interacts directly with the surface water through the processes mentioned in the previous paragraph. Within the bed, additional fate and transport processes include mixing within the surface sediment (due to bioturbation, as well as mixing from physical forces, including vessel propwash); advection and dispersion of porewater throughout the bed (i.e., in surface and subsurface sediment); partitioning; biodegradation (for some chemicals, under certain conditions); and NAPL transport processes, including migration via advection, dissolution, and gas ebullition. Chemicals can also enter the sediment due to the discharge of groundwater from: 1) the underlying native material into the subsurface sediment; and 2) adjacent fill and post-glacial materials, where they are in direct contact with the surface sediment along the shoreline (this pathway is spatially limited[202]).

Net sedimentation, which occurs when deposition of sediment occurs at a greater rate than erosion, is the process by which layers of newly deposited sediment from the water column successively act to "bury" existing surface sediment. As new sediment is deposited atop the existing surface layer, the sediment and associated chemicals within the original surface layer are effectively buried deeper below the sediment/water interface (although some fraction of those sediments are retained in the surface layer through mixing processes). Over time, when the new depositing sediment has lower concentrations than the existing surface sediment, this deposition-driven burial, in conjunction with surface mixing, results in a vertical concentration gradient. That gradient is characterized by lower concentrations at the sediment surface and in shallower sediment, as compared to higher concentrations deeper in the sediment bed, which reflect sediment associated with periods of higher historical chemical loads and concentrations.

The remainder of this section presents evaluations of the Study Area data conducted to develop a better understanding of these fate and transport processes, in order to support the development of the CSM presented in Section 8 and ultimately to guide evaluations of remedial alternatives in

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[202] The surface area associated with the interface of the 15-cm thick surface sediment layer with the adjacent upland materials across the perimeter of the Study Area is less than 1% of the surface area of the contact between the subsurface sediment and the underlying native material.

the FS. In the cases of hydrodynamic and sediment transport processes, results from modeling (as documented in Appendix G) are used to provide further understanding of these processes. A mechanistic CFT model is also being developed to further evaluate these processes for the FS, as described in the MAM2 (Anchor QEA 2016c). Because the CFT model is still in the development stage, the evaluations of CFT processes in this RI Report are preliminary and less quantitative (and will be refined in the future), as compared to hydrodynamics and sediment transport, which are substantively complete. Future refinements to the CSM based on modeling activities conducted after USEPA approval of the RI Report will be documented in future modeling reports or the FS Report.

6.2 Hydrodynamics

A detailed discussion of Study Area hydrodynamics, including analysis of numerous datasets collected in the Study Area and documentation of the development and calibration of a mathematical model, are presented in Section 4 of Appendix G. A summary of the key points relevant to assessing fate and transport is provided in this subsection. The following are the primary findings of the hydrodynamic data and modeling analyses:

- Newtown Creek is a dead-end tidal channel with relatively low current velocities.
- Episodic point source and overland flow discharges of freshwater during rain events generate density-driven circulation, with the strongest salinity stratification occurring in the tributaries, which affects the transport of suspended sediment and chemicals.

6.2.1 Freshwater Inflow

Freshwater from point sources and overland flow discharges into the Study Area both continuously (e.g., treated effluent from groundwater dewatering and remediation systems) and during episodic rain events (i.e., CSOs, WWTP treated effluent overflow, or stormwater). The characteristics of these flow inputs are described in detail in Sections 3 and 4 of Appendix G.

The relative contributions of flow from these sources are discussed in Section 5.1.2, which explains that CSOs account for the largest fraction of total freshwater flow among the point sources. A diagnostic analysis of the 2015 geo-neutral point source model results for the 5-year period of 2008 through 2012 (see Section 3.5 of Appendix G) shows that the four largest CSO outfalls (i.e., English Kills [NCB-015], East Branch [NCB-083], Maspeth Creek [NCQ-077], and Dutch Kills [BB-026]), which discharge to the Study Area during episodic rain events, have the following range of discharge characteristics (see Table G3-3 of Appendix G):

- Annual discharge ranging between 130 and 530 MG
- 40 to 80 CSO events per year (i.e., discharges during rain events)
- Discharge duration of 2 to 5 hours per event
- Peak flow rates ranging between 370 and 1,700 gallons per second

A diagnostic analysis to evaluate the precipitation amounts needed to trigger CSO events for two of the large CSOs (English Kills and East Branch) indicates that approximately 0.1 inch (East Branch) to between 0.2 and 0.3 inch (English Kills) of precipitation is needed for discharge from these two CSO outfalls to occur (see Section 3.9 of Appendix G).

Stormwater (from large outfalls such as MS4s, as well as from smaller outfalls and overland flow) accounts for the next largest fraction of freshwater inflow, as discussed in Section 5.1.2. A strong correlation exists between stormwater discharge and precipitation; this type of discharge is not affected by sewer system hydraulics and controls. Treated groundwater effluent is discharged to Newtown Creek on a continuous basis from three (previously four) outfall locations (see Figure G4-24 of Appendix G).[203] Although groundwater inflow from the native material beneath the sediment bed and laterally from the shallow fill through vertical permeable shorelines[204] is also another continuous source of freshwater to the Study Area (a net discharge of 690 MG per year for areas with net upward seepage; see Section 5.2.1 and Table 5-16), it has minimal impacts on hydrodynamic processes and modeling results (based on diagnostic testing with the hydrodynamic model; Attachment G-E of Appendix G).

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[203] One of these four outfalls is associated with the Buckeye Pipeline Facility; that outfall only discharged from 1987 to October 2012. In March 2015, NYSDEC discontinued the SPDES permit due to the removal of the treatment system (GES 2017; see Section 3.2.8.2.3 for more details).

[204] As discussed in Section 5.2, shallow lateral groundwater discharge has not been empirically characterized; to support the FS, USEPA is planning a study to collect empirical data with the stated objective of reducing uncertainty in the current lateral groundwater discharge estimates.

6.2.2 Current Velocities, Circulation, and Tidal Effects

Various hydrodynamic datasets collected during Phase 1 and Phase 2 (e.g., current velocity, tidal elevation, temperature, and salinity, during both dry and wet weather) have been combined with historical data (e.g., water surface elevation and residual flow rates in the East River and NYCDEP salinity data) and used in conjunction with results of the hydrodynamic modeling study (see Section 4 of Appendix G) to summarize the hydrodynamic conditions within the Study Area. During dry weather conditions (i.e., minimal inflow of freshwater from point source and overland flow discharges), the hydrodynamics in the Study Area are primarily affected by tidal circulation, with minimal vertical temperature and salinity stratification in the surface water (see Figure 6-2). Typical of a dead-end tidal channel, current velocities have a maximum value near the mouth of Newtown Creek and decrease with increasing distance from the East River, with the lowest current velocities occurring in the upper portions of the Study Area (e.g., East Branch and English Kills). Current meter data show that peak dry weather current velocities range from 0.6 to 0.8 foot per second in CM 0 – 1 to less than 0.2 foot per second in English Kills.

During wet weather conditions when freshwater is discharged into the Study Area from point sources and overland flow, stratified flow conditions can develop, with less-dense, fresher water flowing toward the East River in a surface layer, and denser, more saline water flowing inland in a bottom layer (see Figure 6-3; see also Section 4.7.2.1). This type of hydrodynamic condition is referred to as density-driven circulation. This characteristic of stratification within the Study Area is evident in the comparison of surface/deep salinity samples from dry and wet weather surface water sampling data presented in Section 4.7.2.1. The strongest salinity stratification during point source discharge events occurs in the tributaries and to a lesser degree in CM 2+, with the least amount occurring in CM 0 – 2. Tidal mixing causes the effects of point source discharges on hydrodynamic circulation in the upper portions of the Study Area to decrease with distance, as the point source effluent is transported downstream toward the East River. Due to the relatively short duration of a point source discharge event (e.g., typically on the order of 2 to 6 hours), stratified conditions (i.e., low salinity in the surface layer and high salinity in the bottom layer) typically exist for less than 24 hours within the Study Area. However, water and solids discharged from point sources can remain in the water column for 24 hours or longer, due to mixing and dispersion processes. For example, a 1.5-day tracer simulation was conducted with the hydrodynamic model during a precipitation event in which 1.8 inches of rain fell, and discharges from the four largest CSOs lasted 4 to 6 hours (see Attachment G-E of Appendix G). The results from this simulation showed that predicted tracer concentrations in the upper portions of the creek remained at 30% of the CSO release concentration for more than 12 hours following the event and remained at 10% to 20% of the release concentration at the end of the 1.5-day simulation. This result is consistent with bacteria sampling data collected by NYCDEP as part of the LTCP, which also showed that effects of point source discharges are observed for several days following wet weather events (NYCDEP 2017).

Current meter data show that peak wet weather current velocities range from 0.6 to 0.8 foot per second in CM 0 – 1 and are less than 0.2 foot per second in English Kills. This range of wet weather current velocity data is similar to that for dry weather conditions presented in this subsection. The reason for this similarity is that the distance between the current meters and the CSO/point source outfalls was large enough to result in negligible differences in tidal current velocity data (i.e., 10-minute [raw data] and 3-hour low-pass-filter [tidal component] velocities). Note that differences are observed between dry and wet weather conditions for residual current velocity data (i.e., 34-hour low-pass-filter velocities), with wet weather residual current velocities being greater than those for dry weather, as expected.

6.3 Sediment Transport

A detailed discussion of Study Area sediment transport processes, including analysis of the datasets collected in the Study Area, empirical mass balance calculations, and documentation of the development and calibration of a mathematical model, is presented in Section 5 of Appendix G. A summary of the key sediment transport points relevant to assessing fate and transport is provided in this subsection and in Figure 6-4. The following are the primary findings of the sediment transport data and modeling analyses (see Figure 6-4):

- **CM 0 – 2**
  - Primary sediment source from East River
  - NSRs decrease with increasing distance from the East River
  - Minor changes in NSRs during the last 50 to 75 years
  - Propwash sediment resuspension occurs

- **CM 2+**
  - Sediment source from a combination of CSOs, stormwater, and East River
  - Large decreases in NSRs during the last 50 to 75 years due to reductions in CSO sediment loads and decreases in trapping efficiency
  - Propwash sediment resuspension occurs
- **English Kills, East Branch, Maspeth Creek, and Dutch Kills tributaries**
  - Primary sediment source from CSOs and stormwater
  - Large decreases in NSRs during the last 50 to 75 years due to reductions in CSO sediment loads and decreases in trapping efficiency
  - Propwash sediment resuspension occurs in the lower portion of English Kills

6.3.1 Sediment Bed Characteristics

The sediment bed throughout Newtown Creek is characterized as a cohesive (muddy) bed, with varying amounts of fine (clay- or silt-size particles) and coarse (sand-size particles) material (see Section 5.2.3 of Appendix G). Newtown Creek is primarily a net depositional system, due to low current velocities, although modeling and data indicate that erosional areas (be they episodic or longer term) do exist in the Study Area due to vessel propwash and local impacts of point source discharges.

6.3.2 Sediment Sources and Inputs

Sediment in the Study Area originates primarily from two sources: 1) the East River; and 2) CSO and stormwater discharges (see Section 5.4.3 of Appendix G). Each of these is discussed in this subsection.

The East River is a primary ongoing source of sediment to the Study Area, with sediment being transported into Newtown Creek during each flood tide. Average TSS concentrations measured in the East River near the mouth of Newtown Creek from the Phase 2 sampling in 2014 to 2015 averaged about 30 mg/L (see Section 4.7.2.3). The grain size distribution of solids entering the Study Area from the East River, based on the sampling conducted under Part 1 of the FS (see Section 5.4.3 of Appendix G), is fine-grained (i.e., in the silt- and clay-size range), with components that have varying settling characteristics, including a fraction with very low to near-zero settling (i.e., washload). East River sediment transported into the Study Area mixes with sediment from point source discharges in the water column, with some portion of the East River sediment load (on an annual basis) being deposited within the Study Area (see Section 5.6.1.2 of Appendix G). The East River is the primary source of sediment that is deposited within CM 0 – 2 in Newtown Creek, contributing solids that are mainly composed of fine particles with relatively lower TOC content (see also Section 4.2.2.3). The TOC content of East River solids is generally in the range of 3 wt% to 5 wt%, based on surface sediment spatial distributions (see Section 4.2.2.3) and Phase 2 East River surface water data (see Section 4.7.2.2).

The sediment bed upstream of approximately CM 2 typically contains a mixture of fine and coarse particles with relatively higher TOC content, largely originating from point source discharges (see Sections 4.2.2.3 and 6.3.4) but that are also affected by influences from historical sources (both municipal and industrial). Sediment loads from point source discharges tend to have relatively higher TOC content (i.e., greater than 10 wt% TOC for CSO discharges), as discussed in Section 4.2.2.3, and are composed of a mix of fine and coarse particles. Point source sediment loads occur during episodic discharge events that typically last 2 to 6 hours. Surface water sampling during wet weather events showed higher TSS concentrations during the days of discharge (Round 1) as compared to samples collected on the next day after discharge (Round 2), which is suggestive of solids loads entering the Study Area, then dispersing and settling (see Section 4.7.2.3). Density-driven circulation during a discharge event affects the fate and transport of sediment discharged from CSO and stormwater outfalls. Other processes that affect the transport of sediment discharged from point sources include tidal circulation during dry weather conditions and propwash resuspension from ship traffic in the Study Area.

Additional evaluation of TSS data was conducted as part of the sediment transport model calibration, as discussed in Section 5.5.2.3 of Appendix G. The evaluation of TSS spatial gradients presented in Sections 5.2.4 and 5.5.2.3 of Appendix G demonstrated that during dry weather there is minimal to no spatial gradient from the mouth of the creek to the upstream tributaries. This pattern is consistent with a component of the suspended solids from the

East River subject to negligible deposition within the Study Area (i.e., washload).[205] The modeling and analyses of TSS presented in Appendix G includes an evaluation of spatial patterns in chlorophyll-a data within the Study Area and East River, which indicates that there is likely a contribution of suspended solids from internal algal production that accounts for less than 5 mg/L of the observed TSS during summer months (see Section 5.5.2.3 of Appendix G). A contribution of algal solids is consistent with the spatial pattern in the dry weather f_{OC} data and temporal pattern in the dry weather TSS data discussed in Sections 4.7.2.2 and 4.7.2.3, respectively.

Solids discharged from the CSOs tend to form localized areas of accretion referred to as sediment mounds, in the vicinity of the discharge point. Based on differential bathymetry analyses and modeling results, these localized accretion areas may experience episodic scour during a CSO discharge event, which causes resuspension and redistribution of the CSO solids material downstream of the discharge point within the Study Area (see Sections 2.1.4 and 5.6.1 of Appendix G).

6.3.3 Erosion

The Phase 1 and Phase 2 data and modeling results indicate that hydrodynamic processes (i.e., tidal currents, density-driven circulation) generate relatively low near-bed current velocities throughout large portions of the Study Area, which results in minimal to near zero erosion of the sediment bed (with the exception of the sediment mounds discussed previously). An analysis of bathymetric data collected pre- and post-Hurricane Sandy, which included a major storm surge, showed minimal to minor net bed scour in the Study Area over the time between the two surveys. However, the cause of bed scour inferred from the bathymetry change data cannot be specifically determined, so it is possible that propwash resuspension also contributed to the limited bed scour that was observed.

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[205] On average, dry weather TSS concentrations are similar to those for wet weather. However, as discussed in Section 4.7.2.3, there is a large degree of variability in both datasets, and this comparison may be confounded by the sampling programs being conducted at different times of the year. For example, creek-wide TSS concentrations could have been lower than average conditions prior to the discharges during the wet weather sampling. Regardless, the decrease in wet weather TSS concentrations between Round 1 and Round 2 of sampling shows direct evidence of the solids loads associated with point source discharges.

Propwash from ship movement causes episodic bed scour, mixing, resuspension, and redeposition within the navigation channel, Turning Basin, and lower English Kills (see Sections 5.3 and 5.5.3 of Appendix G). Propwash resuspension is a significant process, affecting sediment transport over a range of spatial scales (i.e., distances ranging from less than 30 feet to greater than 1,000 feet) in the Study Area. Bed scour caused by propwash and subsequent transport of resuspended sediment depends on the following:

- Ship characteristics (e.g., ship draft, propeller type and dimensions, and applied horsepower)
- Location in the Study Area (i.e., water depth)
- Type of ship movement (e.g., cruising in the navigation channel, maneuvering near waterfront facilities or in the Turning Basin, and inbound or outbound)
- Hydrodynamic conditions when bed scour occurs (e.g., ebb or flood tide, high or low tide, dry or wet weather)
- Time period over which propwash impacts occur (i.e., ship speed or passage length)
- Erosion properties of sediment bed

In general, ship propwash effects account for net erosion on small spatial scales (e.g., isolated scour holes). On large spatial scales, propwash effects represent a general mixing/reworking process of the surface sediment (e.g., in CM 0 – 1; see discussion of high-resolution core data in Section 6.4.4.4 for more on this topic). The effects of propwash resuspension have been incorporated into the sediment transport model (see Sections 5.3 and 5.5.3 of Appendix G). A general conclusion of the modeling study was that including propwash resuspension in the sediment transport model improves the performance of the sediment transport model within all areas of Newtown Creek where ship traffic occurs (see Section 5.5.3.5 of Appendix G). Without propwash resuspension, the sediment transport model is not consistent with data-based NSRs at smaller spatial scales (e.g., 0.1-mile-long reaches). Thus, the modeling results show that propwash resuspension is important for understanding NSRs at smaller spatial scales, but not for the larger-scale (e.g., reach average) NSRs within the Study Area.

6.3.4 Deposition and Net Sedimentation

The NSR represents a quantitative metric that integrates the net effects of the various sediment transport processes described previously in this subsection (i.e., sediment loads from East River and point source discharges, erosion, deposition, propwash resuspension, and the sediment trapping efficiency of the waterbody). Estimates of Study Area NSRs vary from approximately 0.5 to 7 centimeters per year (cm/year) across the various methods, time frames, and reaches evaluated. This range of estimated NSRs was based on a multiple LOEs approach, utilizing a range of datasets (i.e., geochronology cores, 1991/1999/2012 bathymetry, and historical dredging information) from the Study Area (see Section 5.2.1 of Appendix G and Figure 6-5). Attachments G-G and G-H of Appendix G provide detailed discussions of the geochronology and differential bathymetry analyses, respectively; these were used to develop the quantitative NSR estimates. For the 10- to 25-year time horizon period (i.e., between approximately 1990 and 2010 to 2015), NSRs are highest near the mouth of the creek (where solids primarily from the East River are deposited), relatively high in the tributaries (where primarily point source solids are deposited), and lowest in the Turning Basin (where solids from a mix of sources are deposited). Relatively low NSRs occur in the Turning Basin, because this area is generally farthest from the primary sediment sources (i.e., East River and tributary point sources). Historical NSRs (50- to 75-year time frame) were higher in the tributaries based on values evaluated using cesium-137 (Cs-137) activity data and dredging records (see Figure 6-5). More recent LOEs representing the 10- to 25-year time horizon suggest that NSRs in the tributaries have decreased over time, due to reductions in solids loads from CSOs and changes in trapping efficiency since the 1960s. The most recent information (1999 to 2012 bathymetry, as well as the high-resolution core chemistry data; see Section 4.3.4.2) indicates that deposition is ongoing throughout the Study Area, albeit at a lower rate than occurred historically. Thus, historical sediment deposits continue to be buried by new sediment from the East River and point source discharges.

Spatial variations in the relative contributions of sediment loads from the East River and point source discharges on deposition rates in the Study Area are shown in Figure 6-6. These results, which are presented as area-weighted averages by reach, are based on sediment transport model predictions (see Section 5.6.1.2 of Appendix G). Within CM 0 to 0.5, East River solids compose approximately 99% of the deposited sediment. The impact of the East River solids on deposition decreases with increasing distance from the creek mouth. East River solids compose approximately 95% of the deposited sediment in CM 0.5 to 1 and CM 1 to 1.5. That percentage decreases to approximately 75% to 85% in Dutch Kills, Whale Creek, and CM 1.5 to 2, then to approximately 65% in CM 2+. East River solids compose 15% to 30% of the deposited sediment within English Kills, East Branch, and Maspeth Creek. Some of the reaches, notably the upstream tributaries, contain smaller-scale spatial variations in the relative contributions of deposited solids that are not apparent in the area-weighted reach averages. This variation is characterized by the fraction of East River solids in the downstream portion of the tributary being relatively higher than that in the upstream portion.

Figure 6-7 provides a condensed summary of sediment transport conditions within the Study Area. This summary was developed using a combination of sediment transport model results and a range of Phase 1 and Phase 2 datasets (e.g., bed grain size distribution, geochronology cores, and bathymetry; see Section 7.3.2 of Appendix G), as well as spatial patterns of bed TOC discussed in Section 4.2.2.2.

6.4 Chemical Fate and Transport Processes

This subsection provides a discussion and evaluation of the CFT processes identified in Section 6.1, including partitioning; surface water transport and mixing; deposition; surface sediment mixing, porewater exchange, and advection; deeper subsurface sediment processes (including effects of groundwater loads); chemical reactions/degradation; and NAPL fate and transport processes. As noted previously, CFT modeling, which is underway and will be completed prior to submittal of the FS, will further quantify these processes.[206] Development of the CSM (see Section 8), and ultimately evaluation of remedial alternatives in the FS, will factor in the understanding and quantification of these processes.

As discussed in Section 4.12, the evaluations presented in this subsection are provided for TPAH (17), TPCB, and Cu. However, additional insights regarding fate and transport can sometimes be gained by evaluating individual chemicals that make up the totals, given the variations in composition and speciation discussed in Section 4.2.5. Indeed, due to the differences in individual PAH and PCB properties, the CFT model being developed to support the FS will simulate five PAH compounds and six PCB homolog groups, which were

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[206] The general approach for CFT modeling is described in the MAM2 (Anchor QEA 2016c). However, because that document was developed prior to the RI Report, revisions to the modeling approach described in that document may be made based on interpretations of RI data and the CSM presented in this report. The modeling will be documented in a report that is submitted to USEPA prior to submittal of the FS.

selected based on evaluations of composition,[207] as well as one metal. These modeled chemicals are as follows:

- PAHs: naphthalene, pyrene, fluoranthene, benzo(a)pyrene, and C3-naphthalenes (the first four of these compounds are included in both TPAH [17] and TPAH [34], whereas the last one is an alkylated group that is included in TPAH [34] but not TPAH [17]; see Table 4-24)
- PCBs: di- through hepta-PCB homolog groups
- Metal: Cu

Therefore, in some of the subsections that follow, these individual PAH compounds and PCB homolog groups are discussed, along with the totals. Likewise, the speciation of Cu is also discussed (see Section 6.4.6.1).[208]

6.4.1 Chemical Partitioning Characteristics

Partitioning between the dissolved and particulate (e.g., sediment) phases is a key fate and transport process at contaminated sediment sites (e.g., USEPA 2005a). Partitioning affects concentrations of chemicals that sorb to water column solids, concentrations within sediment porewater, and exposure/risk to biota. The remainder of this subsection discusses literature-based and site-specific evaluations of partitioning for PAHs, PCBs, and Cu conducted for this RI Report; these evaluations have demonstrated the following:

- PAHs, PCBs, and Cu partition strongly onto the solid phase in a manner that can be predicted using an equilibrium partition coefficient. However, for Newtown Creek, this process is not best described through traditional OC-based approaches for the organics, due to complexities of sources and forms of OC present in the Study Area.[209]

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[207] The selection of these PAH and PCB compounds for the CFT model was made based on identifying those that contribute substantially to the total mass, as well as those that can be related back to relevant risk-based metrics (e.g., subset of PCB homolog groups that accounts for the most significant fraction of PCB mass and individual PAH compounds that correlate with TPAH (17), TPAH (34), and other risk-based toxicity or carcinogenicity sums).

[208] For comparison, literature-based sorption characteristics for C19-C36, 2,3,7,8-TCDD, and Pb are provided as well.

[209] As discussed in the remainder of this section, the conclusions from the partitioning analyses presented in this RI Report, including the influence of OC on partitioning, are being further evaluated as part of the CFT modeling effort that is being conducted as part of the FS.

- Strong partitioning means that fate and transport of chemicals in the Study Area surface water and surface sediment are strongly dictated by sediment transport processes. Dissolved phases within the sediment account for a relatively small fraction of the total mass when partitioning is strong, which inherently limits contaminant mobility through dissolved phase transport in the sediment.[210]

6.4.1.1 Use of Partition Coefficients

Within an aqueous media, chemicals generally exist in the following three phases: particulate-bound, freely dissolved, and (for some chemicals) DOC-bound (which is generally considered to account for colloidal fractions; Chin and Gschwend 1992).[211] Within the sediment, the latter two phases occur within the porewater. The degree to which chemicals partition to the particulate phase is typically described by chemical-specific equilibrium partition coefficients (i.e., K_d). The equilibrium approach is appropriate when sorption and desorption processes occur rapidly relative to other transport processes, such that the amount of chemical mass in the dissolved and particulate phases reaches a local steady state. The K_d represents the ratio of particulate-bound chemical concentration (on a dry weight basis) to the concentration of the chemical in the freely dissolved phase at equilibrium. For organic compounds, this coefficient is often expressed on an OC-normalized basis (i.e., per unit mass of OC in soil, sediment, or particulate matter; K_{OC}), based on the observation that organic chemicals oftentimes partition mainly to the organic fraction of the solids (e.g., Karickhoff 1984). For organic chemicals, empirical relationships have been found between K_{OC} and the n-octanol/water partition coefficient (K_{OW}, which is widely published), with K_{OC} values varying proportionately with the K_{OW} (e.g., Di Toro 1985). However, K_{OC} also varies depending on the types of OC present at a given site, which is particularly important for the Study Area. For metals, partitioning is further complicated in that pH, redox potential, and speciation play a role in the amount of sorption. Chemicals with higher partition coefficients sorb more strongly to the sediment. Literature-based and site-specific partition coefficients for PAHs, PCBs, and Cu are presented in Sections 6.4.1.2 and 6.4.1.3, respectively.

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[210] Although the site-specific evaluations presented in this section focused on individual PAH compounds, PCB homologs, and Cu, literature indicates that the other chemicals that contribute to risk and were evaluated for nature and extent in Section 4 (i.e., C19-C36, 2,3,7,8-TCDD, and Pb) also exhibit strong partitioning to sediment.

[211] In areas where NAPL is present, chemicals exist in a fourth phase as well (non-aqueous fluid). NAPL fate and transport processes are discussed in Section 6.4.7.

6.4.1.2 Literature-Based Partition Coefficients

PAHs, PCBs, and Cu are expected to partition to the sediment strongly, as evidenced by their physicochemical properties published in the literature. The strength of partitioning varies by individual compound within the PCB and PAH classes. As an illustration of literature-based partition coefficients for PAHs, PCBs, and Cu, log K_{OC} and log K_{OW} values for the organic compounds were compiled from the literature, with log K_{OC} calculated for PCBs from log K_{OW} based on the Di Toro (1985) relationship.[212] Table 6-1 lists literature-derived log K_{OC} values for PCB homolog groups and the 34 PAHs included in TPAH (34) (see Section 4.1.3.2 and Table 4-24) and log K_d values for Cu, along with the literature sources from which they were obtained. For comparison, literature-based log K_{OC} values for C19-C36 and 2,3,7,8-TCDD and literature-based log K_d values for Pb are also included in Table 6-1. The values shown in Table 6-1 range from log K_{OC} of 3.2 to 7.2 for PAHs and from 4.9 to 8.4 for PCB homologs; log K_d values for Cu are in the range of 4 to 5. Similarly, the log K_{OC} value for C19-C36 is 8.6 (with individual compounds in this class ranging from 7 to more than 10),[213] the log K_{OC} value for 2,3,7,8-TCDD is 6.6, and the log K_d values for Pb are in the range of 5 to 6.[214] Chemicals with values in these ranges are generally considered strongly sorptive (Chapra 1997). These K_{OC} values span three or more orders of magnitude among

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[212] $\log K_{OC} = 0.00028 + (0.983 \times \log K_{OW})$

[213] The log K_{OC} value for C19-C36 cited here and used for the RI (8.6) is considered highly uncertain. Because C19-C36 represents a class of chemicals with a range of properties, selection of a single representative value is challenging. Additionally, literature documenting the partitioning behavior of C19-C36 is limited. The document from which this value was derived (Gustafsson et al. 1997) does not include sufficient detail to evaluate the reported K_{OC} value; the value is estimated based on an empirical relationship between K_{OC} and K_{OW} using K_{OW} values for a relatively narrow set of aliphatic compounds (five straight-chain alkanes). Also, the document does not discuss how the relationship was derived or the type of samples used to develop the K_{OW} values used in the relationship. As such, the value of 8.6 was conservatively selected from the low end of the range reported for the compounds within this class of chemicals (i.e., range of 7 to more than 10). Aside from the general characterization of partitioning properties presented in this section, the log K_{OC} value of 8.6 for C19-C36 is used for developing estimated dissolved phase concentrations for mid-depth porewater and groundwater samples (see Sections 4.8.3.2.3 and 4.9.3.3, respectively). Because the selected K_{OC} value was on the low end of the range, it is anticipated that the estimated dissolved phase concentrations reported in those sections could be overestimated as a result of its uncertainty.

[214] Log-based values of K_d and K_{OC} are the convention used, because values often vary over several orders of magnitude across different chemicals. The units associated with K_d and K_{OC} are liters per kilogram (L/kg) dry solids and L/kg OC, respectively. These units are the convention reported in the literature and are used throughout this RI Report.

PAHs and PCBs, so there will be significant differences in sorption behavior among these chemicals, but they will all partition predictably in general accordance with their K_{OC} value.

6.4.1.3 Study Area-Specific Partitioning Characteristics

6.4.1.3.1 Datasets and Analysis Approach

Site-specific partition coefficients are developed using paired samples of particulate and dissolved phase chemical concentrations.[215] For the partition coefficients described herein, the data analyses for all media are rooted in the common assumption of equilibrium between phases, as discussed in Section 6.4.1.1. For the Study Area, data-based partition coefficients were developed using paired samples from the following sampling programs:

- Shallow porewater data (see Section 4.8.2) collected ex situ as part of the triad program (36 locations, 0- to 15-cm [0- to 6-inch] depth) and in situ as part of the groundwater program (17 locations, 0- to 15-cm [0- to 6-inch] and 15- to 30-cm [6- to 12-inch] depths) using passive sampling techniques (SPME for organics and peepers for metals), along with the corresponding bulk sediment data
- Point source program samples for which certain organic chemicals were measured in paired particulate/dissolved samples that were separated via filtration (see Section 5.1.3)[216]
- All surface water and point source samples for metals (see Sections 4.7 and 5.1.3), for which total (whole-water, unfiltered) and dissolved (filtered) concentrations were measured

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[215] The analyses to evaluate and develop site-specific partition coefficients presented in this subsection will be further refined as appropriate during the CFT modeling being conducted to support the FS.

[216] The analysis assumes that partitioning is at equilibrium for point source program samples. The fact that the partition coefficients calculated from the point source samples are very similar to those from Study Area paired sediment and porewater samples (as discussed in Section 6.4.1.3.3) indicates that the assumption is valid, although there is some degree of uncertainty. The time for equilibrium to be reached between the aqueous and solid phases in point source samples (i.e., the travel time for a sediment particle from collection point to discharge point) could be relatively short as compared to other media (i.e., sediment and porewater, which are likely in contact for weeks to months, as discussed in Section 6.4.5.3). Although the time between a precipitation event and discharge of water from a point source outfall generally occurs on the timescale of minutes to hours, the travel time of solids is longer, because some mass of solids that enters the sewer infrastructure is trapped there for varying amounts of time (e.g., in catch basins). Thus, the timescales of sorption and desorption for solids present in point source discharges are unknown, and there are no data available to specifically test the equilibrium assumption for the point source samples.

Using these data, particulate phase concentrations on a dry weight basis were plotted against the corresponding dissolved phase concentrations. Particulate phase concentrations consisted of bulk sediment data, reported point source particulate concentrations (normalized to TSS to convert to dry weight units), or total minus dissolved metals concentrations (normalized to TSS) for surface water and point source samples. Dissolved phase concentrations consisted of porewater SPME data (which quantify the freely dissolved fraction, as discussed in Section 4.8.1), peeper data (for metals), reported dissolved phase point source organics results (which include freely dissolved and DOC-bound phases), or filtered metals concentrations from surface water and point source samples. Due to the role that OC typically plays in partitioning of organics (as discussed in Section 6.4.1.2), this analysis was performed on a dry weight basis, as well as on an OC basis.[217, 218] Performing the analysis both ways allowed for an exploration of whether the traditional OC-based approach for partitioning relationships is applicable to the Study Area, given the aforementioned complexities in the Study Area OC. Plotting paired particulate phase versus dissolved phase concentrations produces a relationship with a slope that is equivalent to K_d or K_{OC} (depending on whether OC is included) if a regression is performed on a linear scale with the y-intercept forced to zero. Because such regressions can be strongly influenced by a limited number of high concentration data pairs, the analysis was performed on a log-transformed basis, with the partition coefficient calculated as the arithmetic average of the log ratio of individual particulate/dissolved concentration data pairs.[219] The concentration pairs used for

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[217] The analysis was performed on an OC basis by dividing the particulate phase results by bulk sediment TOC, or f_{OC} (= POC/TSS) from point source samples. Cu was included in the OC-based analysis for completeness and because some metals can associate with OC more strongly under certain conditions.

[218] As summarized in an October 4, 2016 e-mail, USEPA determined that point source POC data are potentially biased low relative to USEPA split-sample data, but there is insufficient information to determine whether either of the datasets are correct or biased. USEPA recommended, as a path forward, adjusting the POC data for the observed difference and conducting a sensitivity analysis to evaluate the effect of key model outputs (e.g., loading), using both the adjusted and unadjusted POC data. In June 2018, USEPA requested that this analysis be presented in the FMRM (Appendix G) and summarized in the RI Report. However, the point source POC data were not used in the FMRM. Because they are used in the partitioning analyses presented in this section, future refined evaluations of partitioning conducted as part of the CFT modeling in the FS will include the requested sensitivity analysis.

[219] When the underlying regression equation (i.e., particulate concentration equals dissolved concentration times partition coefficient) is log-transformed, it becomes a least squares regression analysis with slope forced to a value of one (i.e., log of particulate concentration equals log of dissolved concentration plus log of partition coefficient). The partition coefficient from this regression equation (i.e., the intercept) is mathematically equivalent to the average log ratio of the individual concentration data pairs.

the log K_d calculation consisted of the chemical concentration of sediment (dry weight basis) divided by the porewater chemical concentration; those for the log K_{OC} calculation were the sediment concentration on an OC-normalized basis divided by the porewater concentration. This approach was used to calculate the log K_d or log K_{OC} from each subset of the Study Area data. The standard error of the calculated log K_d or log K_{OC} and the r^2 value from the regression are also presented to describe the variability of the calculated value.

6.4.1.3.2 Partition Coefficient Results

The results of this analysis are presented in separate figures for porewater/sediment data (PAHs, PCBs, and Cu in Figures 6-8a through 6-8f, Figures 6-9a through 6-9g, and Figure 6-10, respectively); point source particulate/dissolved data (PAHs, PCBs, and Cu in Figures 6-11a through 6-11f, Figures 6-12a through 6-12g, and Figure 6-13, respectively); and surface water filtered/unfiltered data (Cu in Figure 6-14). Due to the variation in K_{OC} or K_d among individual PAH and PCB compounds (see Table 6-1), the results for these classes of chemicals are shown for the totals (i.e., TPAH [17] and TPCB), as well as the individual compounds being considered in the CFT modeling (i.e., di- through hepta-PCB homolog groups and five PAHs: naphthalene, pyrene, fluoranthene, benzo(a)pyrene, and C3-naphthalenes). Similar graphics for the full suite of analyzed chemicals (which include the individual PAHs included in TPAH [34] and Pb)[220] are provided in Attachment A-C of Appendix A.

Based on review of the plotted particulate-dissolved phase relationships and the data-based K_d and K_{OC} values, the following general observations can be made:

- The site-specific K_d and K_{OC} values are generally higher, meaning that the majority of the mass of PAHs, PCBs, and Cu will be present in the sediment-bound phase. The Study Area-specific K_d and K_{OC} values are similar to (or higher than) the literature values presented in Table 6-1 (see Tables 6-2a and 6-2b).
- Particulate phase concentrations are proportional to dissolved phase concentrations across the various media and chemicals evaluated, as expected. The relationships between the two phases are well explained by the partition coefficient regression lines in most cases (i.e., the data generally follow the regression lines), although some

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[220] 2,3,7,8-TCDD and C19-C36 were not measured in porewater.

chemicals exhibit a high degree of variability. Variability is not uncommon in site-specific partitioning data relationships (based on review of similar data from other projects[221]), due to challenges with this type of sampling, such as paired sampling of two media and using methods that effectively separate phases, without introducing artifacts. Complexities in the sediment sorbent phases in the Study Area also add to the variability (see Section 6.4.1.3.4). Based on the degree of scatter in the data and the larger standard error values and smaller (and sometimes negative[222]) r^2 values for the partition coefficients, the relationships are generally more variable for PAHs than for PCBs. Some individual PAHs show relatively weak relationships (e.g., naphthalene; see Figure 6-8b), whereas others show very strong relationships (e.g., C3-naphthalenes; see Figure 6-8c). Finally, some chemicals show a few data points well below the regression line (e.g., five samples from CM 2+ and English Kills for benzo(a)pyrene, as shown in Figure 6-8f) for both the dry weight and OC-based analyses. These samples are from the triad program, but it is not clear that methodological differences relative to the groundwater program explain the apparent difference in these few samples. Complexities in the sediment composition, association of some PAHs with the particulate matter itself, and spatial variations likely explain some of these observations. Additional evaluations will be performed to further evaluate these data during CFT model development efforts.

- The relationships between particulate and dissolved phase concentrations for TPAH (17) and TPCB are generally weaker than those for the individual compounds evaluated, based on the scatter in the data, the higher standard error values, and the lower r^2 values for the partition coefficients (note that low FoD limits the analysis for some chemicals in the point sources data). This is expected due to the differences in partitioning behavior among the individual chemicals, as discussed previously.

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[221] For example, standard errors for calculated site-specific K_{OC} values in the range of 0.1 to 0.3 log units were found in similar analyses conducted for paired porewater-sediment samples from Onondaga Lake (Parsons and Anchor QEA 2012).

[222] A negative r^2 can result when the slope or intercept of a regression line is forced through a specified value, but the data indicate that the slope or intercept is actually significantly different from that value. In this case, forcing the regression line with a slope of one also forces the line to be far from the data in some instances. Therefore, a negative r^2 indicates that the regression line may not be a good fit for the data, or that there may be a subset of samples that exhibit complexities or differences that would cause data points that appear well below or above the regression line.

- The particulate-dissolved phase relationships and goodness-of-fit for the regressions on a dry weight (K_d) basis are similar to, or in several cases, somewhat better than those on an OC (K_{OC}) basis (e.g., the amount of scatter in the data is similar between the two cases or lower for the dry weight approach in several cases; and the standard error values of the partition coefficient and regression r^2 values are generally similar between the two approaches, or somewhat improved for the dry weight approach in several cases). The marginal differences between approaches (or somewhat improved relationships on a K_d basis) are observed for both porewater and point source samples, and for both the individual PAH compounds and PCB homologs. In the case of the porewater data, the K_d-based relationships generally show some small improvements in the r^2 values and mixed results in the standard error values, as compared to the K_{OC}-based approach. For the point sources data, the K_d-based relationships generally show some small improvements in both the r^2 and standard error values, as compared to the K_{OC}-based approach. As discussed previously, there are some PAH compounds that show a relatively weak particulate-dissolved phase relationship, but in these cases, the results are observed for both the K_d- and K_{OC}-based approaches. This overall result, (i.e., that the K_{OC}-based relationship does not reduce the scatter in the data or the variability in the partition coefficient as compared to the K_d-based approach) is not expected based on traditional OC partitioning theory, in which K_{OC} is expected to be a better predictor of partitioning than K_d for organics. If OC was the sole important determinant of partitioning, the OC-based values would have substantially lower standard errors and higher regression r^2 values, which is not observed. Given the multiple forms and sources of OC present in the Study Area,[223] this result indicates that complexities of the POC in the Study Area likely add variability in partitioning relationships, rather than reducing it (which is often the case at other sites). Additional evaluation of this result, including evaluations of the influences of soot carbon and the impact that different forms and sources of OC have on partitioning, are provided in Section 6.4.1.3.4. Nonetheless, the dry weight K_d-based partition coefficients for the organic chemicals evaluated generally provide a

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[223] For example, the CSOs represent a large current source and even larger historical source of OC that likely differs in nature from other OC sources such as East River and stormwater (see Section 3.1.6). Additionally, some of the OC in the system is associated with anthropogenic sources, such as soot carbon, tar-like materials, and petroleum products. See Sections 4.2.2 and 4.3.2 for further discussion.

reasonable means of quantifying the relationship between particulate and dissolved phase concentrations (based on the general agreement between the data and the lines corresponding to the data-based partition coefficient). A comparative evaluation of these coefficients is described in Section 6.4.1.3.3.

- The particulate versus dissolved phase concentration relationships for Cu show significant variability, across each of the three media evaluated (porewater, point sources, and surface water, for both dry weight and OC bases). In some cases, the concentration data are within a very limited range (e.g., the dissolved phase surface water concentrations are all within the same order of magnitude; see Figure 6-14). This makes evaluating these relationships difficult, because smaller scale sources of variability (such as analytical variability) can mask the true relationship. Also, metals partitioning is affected by numerous other factors, such as speciation, pH, and redox conditions. One potential effect of differences in such geochemical factors is the observation from the surface sediment and surface water Cu partitioning relationships (see Figures 6-10 and 6-14) that data from CM 2+ are generally above the regression lines representing the average log K_d, indicating stronger partitioning to solids in this reach as compared to other reaches. Overall, the average log K_d values for Cu are similar among the three media evaluated (see Table 6-2b), ranging from 4.6 (point sources and surface water) to 5.0 (porewater)—values that are well within the typical literature range shown in Table 6-1 and are indicative of the strong binding nature of Cu in the Study Area. This finding is consistent with the Σ SEM – AVS evaluations of metals speciation presented in Section 4.2.5.3, which show that sulfides are abundantly present in the Study Area sediment, so mobility and bioavailability of Cu and other metals are limited.

6.4.1.3.3 Relationships with K_{OW} and Effects of Dissolved Organic Carbon Phase

As a means of summarizing the results from the analyses for organic compounds, the regression-derived K_d and K_{OC} values were plotted against literature-based K_{OW} values (see Figures 6-15a through 6-15d).[224] To increase the robustness of this evaluation, these figures include all individual PAH compounds quantified by the analytical method used for the RI program[225] and all quantified PCB congeners, as well as pesticides. Pesticides were included to further demonstrate that the relationships observed span a wide range of K_{OW} values and classes of organic chemicals. Values derived from shallow porewater and point source samples are combined in these figures for comparison. The following observations can be made from these figures:

- K_{OC} follows the well-documented proportional relationship to K_{OW} (see Section 6.4.1.2), except for the values from point source samples at log K_{OW} of greater than approximately six (this deviation is most noticeable for PCBs). Notwithstanding this exception for point source samples (which is explained later in this subsection), the proportional relationship between K_{OC} and K_{OW} occurs across wide ranges of K_{OW}, and for each of the chemical classes evaluated, although some exhibit more variability (e.g., alkylated PAHs), whereas others exhibit very little (e.g., PCBs).
- K_d is similarly proportional to K_{OW}, with similar or even less variability around the linear relationship relative to that for K_{OC}. This further illustrates that analyzing partitioning relationships on an OC basis does not appreciably reduce uncertainty, due to Study Area-specific complexities in OC.
- The K_{OC} values are higher than K_{OW} by factors of about 0.2 log units for PCBs to 1.5 log units for PAHs. These differences are greater than traditional literature-based K_{OC}-K_{OW} relationships (e.g., Di Toro 1985), but are consistent with a wide body of literature indicating that other forms of OC (e.g., soot carbon) bind these compounds

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[224] In these figures, each plotted data point represents one compound: the log regression-based partition coefficient calculated from the Study Area data (point sources and porewater averaged separately and presented in different colors; see Tables 6-2a and 6-2b) and the literature-based K_{OW} value. This is in contrast to Figures 6-8a through 6-14, where each figure represents a single compound, and each data point represents a single pair of bulk/particulate and dissolved measurements.

[225] The PAHs quantified by Method SW8270D-SIM consist of the compounds included in TPAH (17) and TPAH (34) (see Section 4.2.5.1 and Table 4-24), as well as several other alkylated and non-alkylated compounds.

more strongly than detrital OC at field sites (e.g., Jonker and Koelmans 2002). These observed differences in partition coefficients further underscore the complexities of OC-based partitioning in the Study Area, which are discussed further in Section 6.4.1.3.4 (and will be further evaluated as part of the CFT modeling).

The "flattening" behavior of the K_{OC} and K_d values from point source samples at log K_{OW} greater than approximately six (most evident for PCB congeners; see Figure 6-15c) is characteristic of DOC effects (e.g., Chiou et al. 1986). This behavior is not unexpected, because the sampling method (filtration) did not separate freely dissolved and DOC-bound phases in these samples. This behavior is not observed in the porewater data, which is also expected because the SPME sampling method is generally considered to measure only the freely dissolved phase (see Section 4.8.1; USEPA 2012d). To account for the DOC effects in the point source samples, the evaluations were repeated for these data by correcting for the DOC phase using the standard approach documented in partitioning literature (e.g., Burkhard 2000). This approach involves inclusion of terms for the DOC concentration and the DOC partition coefficient, which is often treated as proportional to the chemical's K_{OW}, but can vary (Enell et al. 2016).[226] DOC effects can be important in cases where DOC is present at high concentrations, because it increases the fractions present in the more mobile dissolved phases. Accounting for DOC effects can also improve predictability of partitioning relationships (i.e., for modeling) in cases where DOC is variable. The DOC-corrected evaluation results are shown in Figures 6-16a through 6-16d.

The results of the DOC-corrected K_d and K_{OC} values for the point source samples (see Figures 6-16a through 6-16d) show the expected linear relationship with K_{OW} for all classes of compounds. As with the previous figures (e.g., Figures 6-15a through 6-15d), the K_d results show equivalent, or sometimes reduced, variability relative to the K_{OC} results. The flattening behavior for point source samples with log K_{OW} greater than approximately six is not

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[226] The reported dissolved concentration was corrected for DOC effects by dividing by the term ($1+DOC^*\alpha^*K_{OW}$), where α is an empirically derived parameter. For this analysis, an α value of 0.05 was used, because it results in the best linearization of the resulting K_{OC}-K_{OW} relationship. This value is similar to the value of 0.08 reported by Burkhard (2000) and is somewhat lower than the values reported by Friedman et al. (2011; range of 0.1 to 0.4), who state, "Results also imply that site-specific PCB K_{DOC}s are superior to those calculated with generic [relationships]." This acknowledgment provides support for using a site-specific value (i.e., 0.05) in lieu of literature values.

observed after this DOC correction has been applied (including for PCB congeners). Additionally, the point source results generally match the porewater-derived values very well, for both K_d and K_{OC} and across the various organic chemical classes evaluated. Given the very different media sampled using very different techniques, this similarity demonstrates these values provide a robust representation of Study Area-specific partitioning characteristics.

6.4.1.3.4 Role of Organic Carbon and Soot Carbon

The evaluations described in Sections 6.4.1.3.2 and 6.4.1.3.3 indicate that the Study Area-specific data show partitioning relationships that generally conform to expected literature behavior, which includes the following:

- Strong binding to sediment
- K_{OW} dependence of partition coefficients
- DOC effects for organics at high K_{OW}

Additionally, the relationships show consistency between porewater and point source sample-derived coefficients. However, these data also indicate the role of OC in determining partition behavior is complex within the Study Area—OC does not reduce variability in the partitioning relationships in the Study Area (due to dry weight relationships being similar to [or stronger than] OC-based relationships). This conclusion is similar to that found by Hawthorne et al. (2007) for PAHs across eight historically contaminated sediments; the authors observed that "the common practice of describing partitioning behavior using K_{OC} […] rather than K_d, however, did little to reduce the range in partition coefficients." This result is likely due to the presence of multiple sources and forms of OC. A component of OC that has been implicated in enhanced partitioning is soot carbon.[227] Partitioning of organic

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[227] Soot carbon is generally considered to be a subset of black carbon (Shrestha et al. 2010), but the terminology is often ambiguous (Gustafsson et al. 1997; Schmidt et al. 2001), and the same analytical method is often used for both (e.g., Gustafsson et al. [1997] for soot carbon and Accardi-Dey and Gschwend [2003] for black carbon). In this report, the term "soot carbon" is used exclusively, based on its use in the analytical method documentation, but literature regarding black carbon is also relevant.

chemicals to soot carbon has been shown to be stronger than partitioning to natural OC[228] (Ghosh et al. 2003), and even a relatively small percentage of soot carbon can appreciably affect overall partitioning at a site (Koelmans et al. 2006). The partitioning of PAHs and PCBs to soot carbon is considered additive to the partitioning to natural OC (Accardi-Dey and Gschwend 2003; Hawthorne et al. 2011; Arp et al. 2011; Werner et al. 2010). Following this conceptual model, partitioning of a chemical to the solid phase should be elevated, relative to literature-based predictions, where soot carbon is present, and should also increase as the soot carbon content increases.

Despite the utility of the soot carbon approach at some sites (e.g., citations reviewed in Cornelissen et al. 2005), it is not always successful. For example, Hawthorne et al. state, "Utilizing a two-carbon model incorporating anthropogenic BC [i.e., soot carbon] did not improve predictions over a one-carbon TOC model" (Hawthorne et al. 2011). At complex sites, the failure of soot carbon to improve partitioning relationships may be due to competition for sorption sites from DOC and residual oil (Jonker et al. 2004; Koelmans et al. 2009), sorption differences between pyrogenic and petrogenic PAHs (Thorsen et al. 2004; Jonker and Koelmans 2002), inconsistencies in soot carbon measurements (Jonker and Koelmans 2002), and the presence of other forms of carbon (Yu et al. 2018).

Illustrations of the dependence of partitioning to OC and soot carbon for the 12 chemicals listed in Table 6-2a (i.e., the chemicals included in the CFT model), including TPAH (17) and TPCB, are shown in Figures 6-17a through 6-19. In each plot, the first panel shows the relationship between observed K_d and TOC for each individual sample. If partitioning is strongly associated with OC, the data on this panel should exhibit an upward trend.[229] The second panel shows the equivalent K_{OC} value versus TOC. If partitioning is largely controlled by OC, the data should follow a horizontal line (i.e., exhibit no trend with TOC, because the

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[228] Natural organic matter is hypothesized to be relatively amorphous and the result of decomposition of living biomass, whereas soot carbon is hypothesized to be more rigid and condensed, similar to coal (Burgess and Lohmann 2004). Given the historical terminology, soot carbon, although sometimes "natural" (e.g., a result of forest fires), is not considered part of "natural OC." Based on the analytical methods employed, soot carbon is a subset of the TOC measurement; hence, to estimate natural OC, the soot carbon measurement is subtracted from TOC.

[229] Given that the K_d values are log transformed, the upward trend would generally be curved instead of linear, but should be visible in cases where OC influences partitioning.

y axis is already OC-normalized). The third panel shows the relationship between observed K_d and soot carbon. Analogous to the first panel, if the partitioning is strongly associated with soot carbon, the data should exhibit an upward trend.[230] For the organic chemicals, these figures indicate substantial overall variability in the data, with relatively weak relationships between K_d and TOC for PAHs, and somewhat stronger relationships between K_d and TOC for PCBs (but overall variability still dominates the data). Also, the TOC correction (second panels) suggests a possible overcorrection of K_{OC} (due to negative linear regression slopes). There is no consistent dependence on soot carbon (third panels).

The interpretation of why OC does not significantly reduce variability in the partitioning relationships in the Study Area relates to the variety of OC sources (sorbents) in Newtown Creek and how they vary spatially in the Study Area. The OC sources include natural sources (such as terrestrial and algal), as well as anthropogenic sources, which include discharges of industrial wastewater, sewage, and point sources, and releases of tar-like materials and petroleum products. It is likely that a complex mixture of natural organic matter and anthropogenic OC sources, including soot carbon (and related materials) and weathered residual NAPL, all serve as sorbents to some extent in Newtown Creek, with none dominating partitioning; therefore, they individually cannot be used to improve predictions of partitioning behavior.

The possible effects of forms of OC on partitioning, including soot carbon, as well as other potential explanatory factors such as the role of petroleum-derived carbon sources, will be further evaluated as part of the CFT modeling effort in support of the FS. In particular, the approach taken will depend on that which provides the best model calibration to data.

6.4.1.3.5 Approach for RI Report Evaluations

For the purposes of this RI, dry weight-based partition coefficients (accounting for DOC effects) provide an accurate and reasonable predictor of particulate/dissolved phase relationships, so they were used in subsequent evaluations presented in this RI Report.

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[230] Because partitioning to soot carbon is typically described using a Freundlich isotherm and is considered additive to partitioning to OC, a panel of K_{OC} versus soot carbon (similar to the second panel) cannot be constructed. In any event, the third panel satisfactorily illustrates how partitioning varies with soot carbon.

These K_d values are summarized in Table 6-2b. Elsewhere in the RI Report, these data-based K_d values were used for estimation of particulate and dissolved phase concentrations for surface water and point source samples (see Attachment E-C of Appendix E) and for estimating dissolved phase concentrations of groundwater samples (see Section 3.7.2.1 of Appendix F). This approach will be further evaluated and may be refined as part of the CFT modeling effort that is part of the FS.

6.4.2 Surface Water Chemical Fate and Transport Processes

This subsection presents analyses and interpretations of the Study Area data to evaluate the role of CFT processes, and specifically the effect of the East River and point source discharges, on surface water chemical concentrations. As discussed in Section 6.1, several fate and transport processes contribute to the observed differences in surface water chemical concentrations described in Section 4.7 (i.e., spatial, temporal, and dry versus wet weather). Specifically, chemical concentrations in the water column represent an integration of the following: external sources (i.e., point sources, lateral groundwater flow, shoreline seeps including NAPL seeps, and transport from the East River under flood tide conditions); chemical flux to and from surface sediment via sediment/porewater exchange processes, as well as deposition and propwash resuspension; and transport with tidal currents, including mixing and dilution from East River inflows. This section provides additional discussion of these processes as they relate to the observed patterns in surface water chemical concentrations—some discussion of observed patterns in surface water data was provided previously in Section 4.7. Similar to the presentation of data in Section 4.7, dry and wet weather processes are discussed separately, due to the differences in hydrodynamics and sediment transport under these conditions (see Sections 6.2 and 6.3).[231]

6.4.2.1 Dry Weather

Based on the evaluation of surface water data presented in Section 4.7.3 and the source evaluations presented in Section 5, there are two primary sources of chemicals to the water

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[231] Except where otherwise stated, surface water chemical concentrations discussed in this subsection are based on whole-water (i.e., unfiltered) sample results. Some discussions in subsequent subsections make reference to calculated particulate or dissolved phase concentrations based on the methods described in Attachment E-C of Appendix E.

column in the Study Area under dry weather conditions (i.e., periods when stormwater and CSOs are not normally discharging)[232]:

- Tidal exchange with the East River
- Flux of chemicals from the sediment bed (via surface porewater advection and diffusive and tidally driven exchange)

If the flux from the sediment within the main stem was the dominant fate and transport process affecting the surface water, water column concentrations would be expected to exhibit strong spatial gradients and be much higher relative to those measured in the East River. However, this is not evident in the dry weather data. In fact, spatial patterns of water column TPAH (17), TPCB, and Cu under dry weather conditions show concentrations are relatively constant in the main stem from CM 0 – 2 and are similar to the range of concentrations measured in the East River water column. This is shown in Figures 4-155, 4-163, and 4-169, respectively, which show that the interquartile range boxes for East River concentrations generally overlap the interquartile range boxes for CM 0 – 1, CM 1 – 2, CM 2+, and the lower tributaries.[233] This indicates the influx of East River water strongly affects water column concentrations within most of the main stem and the lower tributaries.

Although there is overlap in the range of concentrations, there is an indication of an increase in dry weather surface water TPAH (17) and TPCB concentrations in the upstream tributaries (specifically English Kills) relative to downstream reaches (see Section 4.7.3 and Figures 4-155 and 4-163). This increase suggests that fluxes from sediment and porewater may be significant drivers of surface water concentrations for these contaminants in the

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[232] As discussed in Section 5.1.4, treated effluent from groundwater remediation and dewatering systems discharges to the Study Area under dry weather conditions. Treated groundwater effluent discharges are the only potentially significant point sources that occur during dry weather and are generally minor contributors to the total load from point sources (i.e., generally less than 1% of the total point source load). One exception to this is the groundwater effluent discharge at CM 0.5 (Con Edison – 11th Street Conduit [DAR No. 110]), which represents the largest TPAH (17) point source load, although the flow rates have not been measured, so the estimated load is uncertain (see Section 5.1.2). Dry weather surface water concentrations measured in this area do not show a signal from this load (see Section 4.7.3.1), likely due to the large amount of dilution from the East River in this area.

[233] Data for TPAH (34), 2,3,7,8-TCDD, and Pb presented and discussed in Section 4.7 show similar patterns (although a large portion of the data for 2,3,7,8-TCDD and Pb were non-detect). C19-C36 was not measured in surface water samples.

upstream tributaries. In contrast to TPAH (17) and TPCB, dry weather surface water column concentrations for Cu are relatively constant throughout the Study Area (see Figure 4-169), including the upstream tributaries. The reason for this difference relative to TPAH (17) and TPCB appears to be driven by the fact that Cu has small differences in concentration between surface water and shallow porewater, as compared to TPAH (17) and TPCB, especially in the upstream reaches (these differences are discussed further in Section 6.4.3.1). Thus, for Cu, the influence of fluxes from the surface sediment is smaller relative to the exchange with the East River. Additional evaluation of porewater chemical flux processes, including comparisons of porewater and surface water concentrations, is provided in Section 6.4.3.1, and quantitative load estimates are developed in Section 6.5.1.3.

As noted in Section 5.1.1.2, infrequent municipal sewer bypasses may also contribute to dry weather surface water contamination. Additionally, other sources discussed in Section 5 that could contribute chemicals to the surface water under dry weather conditions include lateral discharge of groundwater through vertical permeable shorelines,[234] spills and releases from overwater activities, and shoreline contaminant seeps. However, as discussed in that section, these processes occur at limited spatial and/or temporal scales, so they are not considered primary sources.

6.4.2.2 Wet Weather

As described in Section 4.7.4, surface water data collected during wet weather conditions show increases in concentrations relative to dry weather (see Figures 4-178, 4-182, and 4-185 for TPAH (17), TPCB, and Cu, respectively, which show median wet weather concentrations can range anywhere from less than 2 to as much as 5 times higher than median dry weather concentrations). This demonstrates that point source discharges represent a chemical mass load to the water column (as further quantified in Section 5.1.4) under wet weather

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[234] Lateral groundwater discharge and shoreline seeps are additional potential sources of contaminants to the surface water during dry conditions (as discussed in Section 6.1); these were evaluated qualitatively using surface water data, as discussed in Section 6.4 of Appendix F. During the development and calibration of the CFT model, chemical loadings from lateral groundwater discharge will be further evaluated through sensitivity analyses, the results of which will be presented and discussed in the CFT Modeling Report and summarized in the FS Report. Moreover, as discussed in Section 5.2, USEPA is planning a study to collect empirical data on lateral groundwater discharge that will support the FS.

conditions. Also, as discussed in Section 6.3.3, there is potential for some localized erosion of sediment within the shallow sediment mounds at the heads of some tributaries—this erosion may also contribute to the observed increase in wet weather concentrations relative to dry weather concentrations. The wet weather data show considerable spatial and temporal variability in these sources, as evidenced by differences in surface water concentrations across the various wet weather sampling events, and differing patterns among chemicals.

Figures 4-179, 4-183, and 4-186 compare measured concentrations of TPAH (17), TPCB, and Cu in surface water collected during the two rounds of sampling (i.e., Round 1 and Round 2) conducted for each wet weather event, respectively. In general, these figures show that concentrations in the upper sampling depth of the water column (which reflects less dense freshwater inputs) were often higher in samples collected during Round 1 (i.e., when point sources were actively discharging) than in Round 2 (the next day, when these discharges generally had tailed off or ceased), particularly for TPAH (17) and TPCB in several tributaries during multiple sampling events. The observed differences between the Round 1 and Round 2 sampling results indicate that the vertical stratification of chemicals discharged from point sources was relatively short-lived, due to vertical mixing and particulate phase settling processes (see Sections 6.2 and 6.3).

However, chemical concentrations measured in Round 2 were still generally higher than the concentrations observed during dry weather conditions at most locations (particularly for TPAH [17] and TPCB; see Figures 4-178 and 4-182, respectively). This suggests that chemical mass contributed to the water column by point source discharges does not flush out of the Study Area over a single tidal cycle, but likely takes several tidal cycles over the course of a few days to return to levels typically observed under dry weather conditions. This observation is consistent with the hydrodynamic tracer simulation results and NYCDEP bacteria sampling data described in Section 6.2.2.

Surface water sampling conducted during wet weather events covered a range of rainfall durations and intensities, which likely contributed to the variability observed in these data. Table 6-3 provides a summary of rainfall duration and intensity for each event in which wet weather surface water sampling was conducted. Figures 6-20, 6-21, and 6-22 show concentrations of TPAH (17), TPCB, and Cu measured during Round 1 for each storm event, by reach (presented as arithmetic average and the corresponding range). These figures show that concentrations in surface water varied among the sampling events, but qualitatively do not show any consistent relationship with rainfall duration or intensity within a given reach. For example, in English Kills, the highest arithmetic average concentration of TPAH (17) in surface water under wet weather conditions was measured during Event 2 (1.8 µg/L), which was a relatively low intensity/long duration rainfall event; the second highest arithmetic average concentration occurred during Event 1 (1.0 µg/L), which was a high intensity/long duration event (see Figure 6-20).

Similar to TPAH (17) concentrations, TPCB and Cu concentrations in surface water do not show much of a relationship with rainfall duration and intensity (see Figures 6-21 and 6-22, respectively). In contrast to TPAH (17), the TPCB concentrations in English Kills were lowest in Event 1 (the highest total precipitation event). This could indicate a potential dilution effect, where relatively lower TPCB concentrations were associated with stormwater-derived point source inputs in that reach during this event, resulting in lower surface water concentrations. However, the data are too limited to draw definitive conclusions.

As noted in Section 6.4.2.1, tidal exchange with the East River is the dominant mechanism controlling surface water chemical concentrations in the main stem of Newtown Creek and the lower tributaries under dry weather conditions. During wet weather, the spatial pattern in surface water chemical concentrations in the Study Area is influenced by chemical loads from the point source discharges (and possibly localized sediment erosion in some locations) in the areas where the inflows are generally highest (e.g., at the heads of the tributaries). This results in generally higher surface water concentrations in the upper portion of the Study Area, beyond what was observed during dry weather, with decreasing concentrations moving toward the East River, as shown in Figures 4-178, 4-182, and 4-185.

The smaller difference between dry and wet weather surface water concentrations at CM 0 – 1, relative to other upstream areas, suggests the tidal exchange with the East River is not the source of the increase in surface water concentrations observed during wet weather conditions in areas further upstream and the tributaries. That is, point source discharges can account for the increased concentrations in the upper Study Area and tributaries, as evidenced by concentrations of point source discharges being generally higher than those in wet weather surface water (see Section 5 of Appendix E). Alternatively, in CM 0 – 1, the large exchange with the East River surface water resulted in lower surface water concentrations in that reach during the wet weather sampling, relative to the upper portion of the Study Area. This difference would suggest that concentrations in the East River during these wet weather events were not higher than those under dry weather, which could be due to the large dilution in the East River, or the timing of discharges to that waterbody relative to those in the Study Area (although a lack of wet weather surface water data collected from the East River adds uncertainty to this interpretation).

In summary, although there is considerable spatial and temporal variability in the wet weather dataset, there is clear evidence of increases in surface water concentrations during wet weather conditions (relative to dry weather). This demonstrates that point source discharges (the largest being from CSOs and MS4s) represent a load of chemical mass to the water column under wet weather conditions, as evidenced by the relatively higher concentrations observed in point source samples. Furthermore, tidal exchange with the East River is the dominant mechanism controlling surface water chemical concentrations in the main stem of Newtown Creek and the lower tributaries under dry weather conditions. This process of tidal exchange is the likely cause of the lower concentrations in CM 0 – 1, as compared to the upper portion of the Study Area observed during wet weather sampling. However, tidal exchange was not the mechanism causing the increase in surface water concentrations observed during wet weather conditions in the upstream portion of the Study Area.

6.4.3 Sediment/Surface Water Exchange Processes

Understanding the exchange of chemicals between sediment and surface water is important to the development of the CSM (see Section 8) and ultimately to remedy evaluation in the FS. As discussed in Section 6.1, several processes can result in exchange of chemicals at the sediment/water interface. Some processes are driven by sediment transport, including sediment resuspension and deposition, and some processes are driven by the dissolved phase (both freely dissolved and DOC-bound phases), including porewater advection (driven by groundwater discharge) and other mechanisms that are termed “diffusive” porewater exchange processes (driven by a concentration gradient between surface porewater and surface water). Diffusive porewater exchange encompasses multiple processes acting in concert, resulting in dissolved phase chemical exchange at the sediment/water interface (e.g., Thibodeaux and Bierman 2003). These processes include diffusion, porewater transfer associated with bioturbation (sometimes referred to as bioirrigation), micro-scale propagation of turbulence, and tidal exchange driven by daily fluctuations in hydraulic gradients and advective flow, including flow direction reversals in some areas (as documented in the USGS reports on seepage meter measurements, which are included in Attachments Bi-B9 and Bii-B1 of Appendices Bi and Bii, respectively).[235]

Sections 6.4.3.1 through 6.4.3.3 include discussions of how RI data shed light on sediment/ water exchange processes in regard to the following:

- Porewater and bulk sediment chemical data and porewater DOC data are used to evaluate the impacts of DOC binding, groundwater flow, and tidal exchange on contaminant concentrations in surface sediment porewater.
- Porewater and dry weather surface water data are used to evaluate porewater advection and dissolved phase exchange.
- Sediment trap and bulk surface sediment data are evaluated to better understand the relative impacts of ongoing deposition from point sources and the East River, compared with localized resuspension (e.g., driven by propwash).

6.4.3.1 Processes That Influence Surface Porewater Concentrations

The concentrations and spatial patterns of shallow porewater data are presented in Section 4.8.2; these data include surface porewater data collected from the 0 to 15 cm interval (0 to 6 inches), and for a subset of locations (17 of the 53 locations), porewater data collected from the 15 to 30 cm interval (6 to 12 inches). As discussed in that section, these data were collected by passive sampling methods (i.e., SPME and peepers), which may introduce limitations (i.e., effects of DOC-bound phase and potential in situ temporal variability, depending on rates of dissolved phase versus sediment phase processes; see Section 4.8.1). Thus, these data were further evaluated to characterize the importance of DOC binding, advection of porewater (driven by groundwater seepage), tidal exchange, and sorption/desorption with the sediment on surface porewater concentrations sampled via

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[235] Gas ebullition may also contribute to dissolved phase exchange of chemicals from sediment to surface water by displacing porewater; that process, as it pertains to NAPL, is discussed further in Section 6.4.7.

passive samplers. The purpose of this evaluation is to support the assessment of dissolved phase exchange between surface sediment and surface water.

6.4.3.1.1 Dissolved Organic Carbon-Bound Phase in Surface Porewater

As discussed in Section 4.8.1, the SPME method used to sample organics in surface porewater only quantitates the freely dissolved fraction of chemicals—it does not quantify the DOC-bound fraction of the chemicals. The importance of the DOC-bound phase is dependent on the concentration of DOC in the surface porewater and the DOC partitioning behavior of each chemical or chemical class (see Section 6.4.1.3). A large fraction (75%) of the surface porewater DOC data from porewater sampling programs was non-detect (see Figure A-A5-8 and Table A-A5-8 in Attachment A-A of Appendix A).[236] When detected, the DOC concentrations were variable and generally low relative to the concentrations at which the DOC-bound chemical accounts for a significant fraction of dissolved phase chemical concentrations; only 1 of the 13 detected samples had a concentration greater than 100 mg/L, with the median value of the detected results being 34 mg/L (and an overall arithmetic average of all results, including non-detects at the MDL, of 15 mg/L). Typically, DOC levels greater than 100 mg/L are needed to strongly affect partitioning for chemicals with K_{OW} less than 10^6 liters per kilogram (L/kg), which applies to most PAHs and many PCBs (i.e., mono- through tetra-chlorinated homolog groups [Hawker and Connell 1988]; these account for the majority of PCBs in the Study Area porewater data). This point is illustrated by Figure 6-23, which shows the ratio of the total dissolved chemical concentration in porewater (i.e., freely dissolved plus DOC-bound) to the freely dissolved concentration (i.e., the case in which DOC is not present), as a function of DOC concentration and K_{OW}.[237] The figure shows that the effect of DOC concentration is less than a factor of two across the range of K_{OW}, from 10^3 to 10^6 L/kg at DOC values less than 20 mg/L (i.e., the range of surface porewater non-detect values). Thus, DOC impacts were not quantified in the evaluations of surface porewater flux in the RI Report. Nevertheless, this phase will be included, for completeness, in the CFT modeling effort being developed to support the FS.

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[236] Ninety-two percent (36 of 39 samples) of the non-detects in the surface porewater were at detection limits of 2.5 mg/L or less, with three samples at detection limits of 10 to 20 mg/L.

[237] The ratio was calculated as a function of DOC concentration based on the equation presented in Section 6.4.1.3.3 (see Footnote 226).

6.4.3.1.2 Effects of Tidal Exchange and Advection on Surface Porewater

As discussed in Section 4.8.1, surface porewater collected by in situ SPME sampling (i.e., from the RI groundwater investigation) can be affected by sediment sorption/desorption kinetics and dissolved phase processes, including porewater advection and tidal exchange. The impacts of these dynamics on in situ porewater concentrations measured by SPME depend on the significance of these dissolved phase processes relative to sediment sorption/desorption. In this evaluation, tidal exchange refers to the downward movement of surface water into the sediment bed due to a downward hydraulic gradient during high tide, with a reversal in gradient and flow direction at low tide. The potential impact of tidal exchange on surface porewater concentrations measured with in situ SPME samplers was investigated in several ways, as discussed in the remainder of this subsection.

First, porewater concentrations measured in samples collected as part of the BERA triad program and the RI groundwater investigation were compared, because differences in sampling methods between these two programs provide a means of assessing the effects of dissolved phase dynamics on the results. The RI groundwater investigation used in situ SPME samplers for organics and peepers for metals (from both 0- to 15-cm [0- to 6-inch] and 15- to 30-cm [6- to 12-inch] depth intervals), whereas the BERA triad sampling used the same types of passive samplers, except the sampling was done ex situ in a laboratory setting (for sediments from the 0- to 15-cm [0- to 6-inch] depth interval). Thus, the groundwater investigation samples were subject to dynamics from tidal exchange and porewater advection during in situ sampler deployment, whereas the triad program samples were not subject to these dissolved phase processes, because passive sampler deployment occurred ex situ in the laboratory. These two sampling programs were conducted at different times (within the same year), and although both programs collected multiple samples from most Study Area reaches, none of the samples from the two programs were collected from the same specific locations (nor were they intended to be). Therefore, comparisons between the data from these two programs need to acknowledge the potential impacts of small-scale spatial variations in concentration (i.e., heterogeneity). Due to these effects, a qualitative comparison of longitudinal concentration profiles for the two programs was performed. These longitudinal profiles show that concentrations of TPAH (17) and TPCB are similar between these two programs (see Figures 4-197 and 4-201), even when accounting for both potential small-scale longitudinal (i.e., upstream/downstream direction) and lateral (e.g., shoreline vs. navigation channel) concentration differences. This similarity between the data from the two programs indicates that the impacts of tidal exchange and porewater advection on observed concentrations were not significant compared to sorption/desorption. For Cu, groundwater program samples from the two depth intervals were similar (see Figure 4-203). This also indicates that dissolved phase dynamics did not strongly affect the measured concentrations, because significant tidal exchange or porewater advection dynamics would have been expected to lead to differences in porewater concentration with depth in the sediment bed, given the magnitude of Study Area porewater velocities (e.g., net seepage rates less than 8.3 cm/day; see Section 5.2.1). The groundwater program porewater concentrations for Cu (both 0 to 15 cm [0 to 6 inches] and 15 to 30 cm [6 to 12 inches]) were generally higher than those from the triad program (0 to 15 cm [0 to 6 inches]), but this could be a result of differences in sampling and laboratory methods, as indicated in Section 4.8.2.2.4. In summary, dominant impacts from tidal effects and porewater advection were not observed. Second, concentration gradients between surface water and shallow porewater provide further evidence that tidal exchange is not a dominant process affecting the in situ porewater samples. Box plots comparing dissolved phase[238] surface water concentrations sampled during dry weather[239] with shallow porewater concentrations are shown in Figures 6-24a through 6-26. Comparisons for TPAH (17) and the individual PAH compounds included in the CFT model are presented in Figures 6-24a through 6-24f; TPCB and the individual PCB homologs included in the CFT model are presented in Figures 6-25a through 6-25g; and Cu is presented in Figure 6-26. These figures show that concentrations of TPAH (17) and TPCB, as well as those of the individual PAH compounds and PCB homologs included in the CFT model, in porewater consistently exceed those in surface water. If tidal exchange dominated porewater dynamics, then similar concentrations would be expected in surface water and shallow porewater. Comparing the individual PAH compounds and PCB homologs, there are variations in the concentration differences observed, with lower molecular weight chemicals showing greater

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[238] Dissolved phase concentrations were calculated based on measured whole water PAH or PCB concentrations and measured TSS and DOC from the corresponding surface water samples using partitioning formulae (see Attachment E-C of Appendix E) and the Study Area-specific K_d values presented in Table 6-2b.

[239] Consistent with the data treatment in Section 4.7, these box plots of dissolved phase dry weather surface water concentrations included samples from both shallow and deep depths (see Section 4.7.1 for the definition of sampling depths). As discussed in Sections 4.7.3.1.2 and 4.7.3.3.2, generally there were no systematic differences observed with depth in the dry weather surface water data.

concentration differences than those with higher molecular weight.[240] These differences likely reflect differences in partitioning behavior within the sediment for PAHs and PCBs (i.e., porewater concentrations are higher for the lower molecular weight, less sorptive chemicals). Additionally, results from preliminary CFT modeling indicate the presence of a loss process in the water column for the PAH compounds (likely a combination of biodegradation and photolysis)[241] that may also be contributing to these differences to some extent. This loss process is being further evaluated as part of the CFT modeling, including conducing sensitivity and bounding analyses. Concentrations of Cu in porewater are generally similar to those measured in surface water; based on the TPAH (17) and TPCB relationships, the cause for this similarity must be due to mechanisms other than tidal exchange (i.e., strong partitioning within surface sediments; see Section 6.4.1.3).

Third, a comparison of calculated log K_d in the 0- to 15-cm (0- to 6-inch) depth interval and the corresponding 15- to 30-cm (6- to 12-inch) depth interval provides further evidence that tidal exchange is not a primary mechanism affecting in situ SPME and peeper porewater concentrations. Consistent with the discussion in Section 6.4.1.3.1, log K_d was calculated for each sample using collocated bulk sediment and porewater concentrations from the groundwater program samples for Cu and the individual PAH and PCB compounds being included in the CFT model.[242] Although there is variability, log K_d values in the 0- to 15-cm (0- to 6-inch) interval are generally similar to or less than in the 15- to 30-cm (6- to 12-inch) interval (see Figure 6-27). In cases where the binomial tests indicate a significant difference ($p < 0.05$), most of the data lie above the one-to-one line, indicating that porewater concentrations in the 0- to 15-cm (0- to 6-inch) interval are higher, relative to sediment

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[240] When comparing median concentrations between shallow porewater and dissolved phase surface water across reaches and the individual PAH compounds and PCB homologs, the following can be observed with respect to the ratios between the two values:

- PAHs: median shallow porewater concentrations are 10 to 20 times greater than those for surface water for the HPAHs (fluoranthene, pyrene, and benzo(a)pyrene), with higher differences (factors in the range 30 to 100) for the LPAHs (naphthalene and C3-naphthalenes).
- PCBs: median shallow porewater concentrations are approximately 5 times greater than those for surface water, with some trend of smaller differences with increasing chlorination level (although this trend is confounded by differing PCB composition in Dutch Kills and Maspeth Creek; see Section 4.2.5.2).

[241] Degradation processes are discussed in Section 6.4.6.3.

[242] Other figures referenced in this section only present results for TPAH (17), TPCB, and Cu, because similar figures for the full suite of chemicals, including the 12 chemicals included in the CFT model, are provided in Attachment A-A of Appendix A.

concentrations, than in the 15- to 30-cm (6- to 12-inch) interval. If tidal exchange dynamics were a dominant driver of observed surface porewater concentrations from the in situ SPME samples, porewater concentrations in the surface 0- to 15-cm (0- to 6-inch) interval would be expected to be lower, relative to sediment concentrations, as compared to the 15- to 30-cm (6- to 12-inch) interval, due to infiltration of lower concentration surface water. This difference would have resulted in higher K_d for the 0- to 15-cm (0- to 6-inch) samples—if anything, the data plotted in Figure 6-27 show the opposite (i.e., the data do not support higher K_d for the 0- to 15-cm [0- to 6-inch] samples).

Finally, to further evaluate the potential effects from porewater advection and tidal exchange together, collocated bulk sediment and porewater concentrations for TPAH (17), TPCB, and Cu were plotted against each other in Figures 6-28 through 6-30. Based on average net seepage rates (see Table F5-10 of Appendix F) and the tidal variations in high frequency gross seepage measurements from the USGS seepage meter studies (see Attachments Bi-B9 and Bii-B1 of Appendices Bi and Bii, respectively), these figures separate porewater sample locations into the following three categories:

1. **Locations where tidal exchange effects are expected to be more significant**. This category was assigned at locations where there was a large observed peak-to-peak amplitude of tidal seepage fluctuations relative to the net average seepage rate, and these fluctuations resulted in gross negative seepage rates.[243] These locations are indicated by red circles in the figures.
2. **Locations where upward advection effects are expected to be more significant**. This category was assigned for cases where the net average seepage rate was relatively high and remained predominately positive, with a tidal signal that was either non-existent

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[243] A location was classified as "Tidal Exchange More Significant" if it met the following criteria: 1) the typical peak-to-peak amplitude of tidal seepage fluctuations was greater than 4 cm/day, which is twice the average of net seepage values from the full dataset (excluding those with net negative seepage); and 2) negative seepage values were measured. The tidal seepage fluctuation peak-to-peak amplitudes were based on figures showing the smoothed 1-minute measured seepage rates presented in Attachments Bi-B9 and Bii-B1-1 of Appendices Bi and Bii, respectively.

or small in peak-to-peak amplitude relative to the net average.[244] These locations are indicated by green circles in the figures.

3. **Locations where tidal exchange and porewater advection are expected to be less significant**. Samples from locations that were not placed into one of the previous two categories were placed in this category; such locations were not circled in the figures.

The results of this categorization at each porewater sampling location from the RI groundwater investigation are presented in Table 6-4. If tidal exchange was the dominant mechanism affecting the porewater concentrations for passive sampling methods (SPME and peepers), one would expect the red-circled symbols to be clustered in the upper-left quadrant of the data relative to the remainder of the dataset (i.e., porewater concentrations would be lower relative to sediment concentrations, due to dilution from lower concentration surface water). If upward porewater advection was the dominant mechanism, one would expect the green-circled symbols to be clustered in the lower-right quadrant of the data relative to the remainder of the dataset (i.e., porewater concentrations would be higher relative to sediment concentrations, because deeper sediment and porewater generally exhibit higher chemical concentrations; see Section 4.8.2.3). For all three chemicals, the results do not show such segregation of the data, providing little evidence that either tidal exchange or porewater advection exert a dominant influence on measured porewater concentrations, and further suggesting that sorption/desorption within the surrounding sediments are the main processes dictating observed porewater concentrations.

The analyses presented in this subsection demonstrate that shallow porewater chemical concentrations are largely controlled by partitioning with sediments (sorption/desorption effects) and that tidal exchange and porewater advection have, at most, secondary influence on the observed concentrations. Also, DOC sorption is not a significant process controlling dissolved phase concentrations. Thus, potential limitations associated with these processes do not appear to affect the usability of the concentrations measured with in situ SPME and peeper samplers. Based on the observed porewater concentrations, evaluations of the processes of tidal exchange and porewater advection as contributing to porewater fluxes at

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[244] A location was classified as "Porewater Advection More Significant" if it met the following criteria: 1) the net average seepage rate was greater than 2 cm/day; and 2) measured gross seepage rates remained positive for most or all of the meter deployment.

the sediment/water interface are discussed further in Sections 6.4.3.2 and 6.5. These process (including DOC-bound transport) will be further evaluated as part of the CFT model being developed in support of the FS.

6.4.3.2 Dissolved Phase Sediment/Water Exchange

As discussed in Section 6.4.3.1, there are chemical concentration gradients between surface water and porewater that are driving "diffusive" and tidal porewater exchange processes at the sediment/water interface (see Figures 6-24a through 6-24f and 6-25a through 6-25g), which along with porewater advection driven by groundwater flow, contribute flux to the surface water. The additional analysis presented in this section provides further evidence of the presence of these dissolved phase fluxes. As discussed in Section 6.4.2.1, the dry weather surface water data presented in Section 4.7.3 indicate that most of the main stem surface water concentrations are consistent with those observed in the East River surface water entering the Study Area. However, increases in surface water TPAH (17) and TPCB[245] concentrations are evident in the upstream tributaries (especially English Kills).[246] This suggests that the flux of chemicals from porewater may be relatively more significant in the upstream tributaries. To further evaluate porewater exchange for PAHs and PCBs, spatial gradients in dry weather surface water data for some individual compounds (i.e., those being included in the CFT model, as discussed previously) are presented in Figures 6-31a through 6-31e for PAHs and Figures 6-32a through 6-32f for PCBs.

As discussed in Section 6.4.1, partitioning characteristics vary among these chemicals—for a given bulk sediment concentration, those that have lower K_{OW} are expected to be present in porewater at relatively higher concentrations, while chemicals with higher K_{OW} are more sorptive and tend to be more strongly bound to particulates. The data shown in

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[245] As acknowledged in Section 4.7.3.3.1, there are lower sample counts of TPCB congener dry weather surface water data. Although there are fewer samples for PCB homologs, the dataset is sufficient to qualitatively evaluate trends, especially because trends observed in the TPCB Aroclor data are consistent (see Figure 4-164).

[246] These increases are generally not observed for Cu. This likely is caused by differences in sorption strength in the surface sediment, as evidenced by porewater data. Specifically, comparing shallow porewater and dry weather surface water concentrations in the upper reaches of the Study Area (see Figures 6-24a through 6-26) shows that the relative difference between shallow porewater concentrations and surface water concentrations is much greater for TPAH (17) and TPCB than Cu. This means that there is much less of a driving gradient for dissolved phase flux for Cu than for TPAH (17) and TPCB.

Figures 6-31a through 6-31e and 6-32a through 6-32f illustrate that compounds with lower Kow (i.e., less sorptive chemicals) have spatial gradients in dry weather surface water that are more evident, as compared to the TPAH (17) and TPCB results. For example, the following differences are observed:

- TPAH (17) concentrations under dry weather (see Figure 4-155) show limited gradient; concentrations in the upper Study Area are slightly higher, but by less than a factor of two on average, relative to the lower Study Area. However, some of the individual PAHs with lower Kow (e.g., C3-naphthalenes; see Figure 6-31e) show a more prominent gradient, with a twofold difference in concentrations between the lower and upper reaches of the Study Area. Note that patterns for naphthalene (another individual PAH with relatively low Kow) are confounded by a greater proportion of non-detects.
- Although TPCB shows a spatial pattern with concentrations that are two to three times higher in English Kills than the rest of the Study Area (see Figure 4-163), the lower homolog groups (e.g., di through tetra; see Figures 6-32a through 6-32c) that have lower Kow values show a larger difference (e.g., three- to fourfold) between English Kills and the rest of the Study Area; and the higher groups (e.g., hexa and hepta; see Figures 6-32e and 6-32f) that have higher Kow values show a smaller difference between English Kills and the rest of the Study Area.

The more prominent spatial gradients for individual compounds with lower partition coefficients, coupled with the observed concentration gradients between surface porewater and surface water, indicate the porewater flux is likely the mechanism responsible for the observed increase in dry weather surface water concentrations in the upper tributaries. The impact of porewater flux on surface water concentrations also needs to account for the hydrodynamics in the system, especially in the upper tributaries where porewater flux can potentially impact surface water in adjacent reaches—thus, spatial patterns in dry weather surface water will not match exactly the patterns of porewater concentration. To better understand the relative impact of the dissolved phase chemical exchange processes on the Study Area surface water, chemical load calculations for surface porewater flux to surface water (from both advection and diffusive exchange) were conducted for comparison to other load mechanisms (see Section 6.5). Further evaluation and quantification of dissolved phase sediment/water column exchange processes will be conducted as part of the CFT modeling that is being developed for the FS.

6.4.3.3 Particulate Phase Sediment/Water Exchange

Sediment traps were deployed to help understand the nature and quantity of solids and chemicals depositing on the sediment bed. As described in Sections 2.1.3.4 and 4.5, the Phase 2 sediment trap study consisted of traps deployed atop the sediment bed at 30 locations in the Study Area (e.g., see Figure 4-100), for three quarterly deployment periods, totaling 9 months (i.e., Q1, Q2, and Q3[247]; see Section 4.5.1 for additional deployment details). The sediment trap concentration data, including observations of trends and variability, are presented in Section 4.5.3; additional evaluation of these data with respect to particulate phase sediment/water exchange is provided in this subsection.

The following discussion focuses on the relationship between chemical concentrations in sediment traps and surrounding surface sediment, as a means of evaluating the source of the material collected in the traps.[248] A series of longitudinal profiles are plotted that show the sediment trap data and surface sediment data together for TPAH (17), TPCB, and Cu (see Figures 6-33 through 6-35).[249] In addition, the data were evaluated using a series of cross plots, which present sediment trap concentrations (arithmetic average of the results from the three sampling quarters) on the vertical axis versus the arithmetic average concentration of surface sediment samples collected within a 0.1-mile radius of the sediment trap on the horizontal axis (see Figures 6-36 through 6-38), along with a linear regression of these data.

Throughout the Study Area, concentrations of TPAH (17), TPCB, and Cu in the sediment traps trend generally positively with the concentrations in nearby surface sediment, as seen on the longitudinal profiles, as well as on the cross plots, which show a positive slope of the regression line for all three chemicals (although the r^2 values in the range of 0.3 indicate there is

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[247] As discussed in Section 4.5.1, Q1 deployment occurred mid-June through mid-September of 2014, Q2 was mid-September through mid-December of 2014, and Q3 was mid-December 2014 through mid-March 2015.

[248] Although it is recognized that sample collection methods differed between these two media (sediment traps versus surface sediment grab samples), where, for example, the sediment trap samples have a higher water content, it is believed that this qualitative comparison is valid.

[249] Sediment trap data for other chemicals considered in the nature and extent evaluation (i.e., TPAH [34], C19-C36, 2,3,7,8-TCDD, and Pb) are discussed in Section 4.5.3.

variability in the relationships related to factors other than local bulk sediment concentration). The positive trend with local surface sediment concentration suggests the material collected in the sediment traps included some amount of solids that were resuspended from the nearby sediment bed, which might be a reflection of vessel propwash effects in many areas.

The scatter in the relationships between surface sediment and sediment trap concentrations (see Figures 6-36 through 6-38) indicates that contribution from local resuspension does not account fully for chemical concentrations in the sediment traps. Other sources of solids (in particular, point source discharges and East River) also contribute to the material deposited in the sediment traps, and other sources of chemicals to the surface water (see Figure 6-1) may also have influenced the concentrations measured in the sediment traps. The relative importance of depositing solids sources varies among the reaches of the Study Area (e.g., see Section 6.3.4). TPAH (17), TPCB, and Cu concentrations in sediment traps collected in both the CM 0 – 1 and CM 1 – 2 reaches trend with (see Figures 6-33 through 6-35), and are generally similar to, concentrations in nearby surface sediment (i.e., the data lie generally along the diagonal 1:1 line in Figures 6-36 through 6-38). For TPCB and Cu, the sediment trap and surface sediment data in these reaches are clustered tightly around the 1:1 line. For TPAH (17), the data in these reaches are also generally clustered around the 1:1 line, although they are slightly more scattered than for TPCB and Cu. The following two factors may contribute to the similarity between sediment trap and nearby surface sediment concentrations observed in CM 0 – 2:

1. **Locally resuspended sediments contribute to sediment traps in the CM 0 – 2 area**. This region experiences the most ship traffic, which likely influences local sediment resuspension and redeposition (see Section 6.3; see also Section 5.3.2 of Appendix G).
2. **Chemical concentrations in surface sediment in the CM 0 – 2 area are similar to those of incoming solids from East River**. Chemical concentrations in the sediment of this reach are similar to those of depositing solids entering from the East River. Specifically, the ranges of estimated and calculated particulate phase chemical concentrations from the East River surface water sampling program (see Section 5.3) generally overlap the ranges of concentrations from CM 0 – 2 surface sediment and sediment trap data (based on comparing the values in Tables 5-21, 5-23, and 5-25 to Figures 6-33, 6-34, and 6-35, respectively). The degree of overlap in these ranges varies somewhat by chemical (there is greater overlap for TPCB and Cu than TPAH [17]); there is greater overlap with data

from CM 0 – 1 than from CM 1 – 2, and the overlap occurs more so at the lower ends of the surface sediment concentration ranges.

There appears to be more variability in the data from the tributaries located within CM 0 – 2 (Whale Creek and Dutch Kills), where some points are clustered tightly around the 1:1 line (e.g., PCBs in Whale Creek, Cu in Dutch Kills), whereas others are more scattered, although data are more limited in these tributaries as compared to elsewhere in the Study Area (see Figures 6-36 through 6-38).

In contrast, data in the CM 2+ area (which includes the Turning Basin) and the upstream tributaries (i.e., Maspeth Creek, East Branch, and English Kills) exhibit larger differences between sediment trap and surface sediment concentrations in some cases, as compared to the CM 0 – 2 data. For TPAH (17), sediment trap data in the upstream tributaries and CM 2+ are more similar to the nearby surface sediment data, as compared to TPCB and Cu (although in general, there are more data points below the 1:1 line than above for each of the three chemicals, as shown in Figures 6-36 through 6-38). For TPCB and Cu, concentrations in some of the traps from these areas are almost a factor of 10 lower than nearby surface sediment (i.e., data points are generally below the 1:1 line in Figures 6-37 and 6-38, respectively). The average TPAH (17), TPCB, and Cu concentrations measured in the traps in CM 2+ and upstream tributaries are generally greater than observed in the traps in CM 0– 2 and downstream tributaries (Whale Creek and Dutch Kills; see Figures 6-36 through 6-38) and are at the lower end of the nearby surface sediment data (see Figures 6-33 through 6-35).

The pattern of lower concentrations of TPAH (17), TPCB, and Cu in sediment trap data versus the nearby sediment bed in CM 2+ and in upstream tributaries (i.e., Maspeth Creek, East Branch, and English Kills) may be due in part to vertical gradients within the sampled surface sediment (15 cm [6 inches]). That is, chemical concentrations in resuspended sediment may be more similar to the trap data than they appear in Figures 6-36 through 6-38, to the extent that concentrations increase with depth within the top 15 cm (6 inches), and only a portion of the top 15 cm (6 inches) is resuspended. In addition to locally suspended and redeposited sediment, concentrations in sediment traps may also be influenced by sources of external solids (potentially from point sources, from the East River, and from resuspended sediment from other reaches) and chemicals (including from shallow lateral groundwater loadings, shoreline seeps including NAPL seeps, and gas ebullition[250]).

Regarding overall trends in the tributaries, there are exceptions for TPCB at one trap location in Maspeth Creek and at one in Dutch Kills, where the arithmetic average trap concentration is approximately 3 (Maspeth Creek) to 6 (Dutch Kills) times higher than that of the surrounding surface sediment. These higher (on average) concentrations are specifically driven by Q2 sample results at each location with concentrations of nearly 30 mg/kg (see Section 4.5.3.4). As discussed in the next paragraph, variation in precipitation and point source discharges, vessel propwash, or other ongoing external sources may explain the differences among quarters.

Concentrations of TPAH (17), TPCB, and Cu in sediment traps were compared among the three quarterly sampling periods, as discussed in Section 4.5.3. Concentrations of TPAH (17) are highest in Q2, lower in Q3, and lowest in Q1 (see Figure 4-110). In contrast, concentrations of TPCB and Cu are, on average, more similar across the three quarters (see Figures 4-119 and 4-125, respectively). Despite the noted temporal differences for TPAH (17), concentrations of all three chemicals in all quarters lie generally within the range of surface sediments throughout the reach in which the traps were deployed and of particulate phase samples of point sources (compare Figures 6-33 through 6-35 [sediment trap and surface sediment longitudinal profiles] with Appendix E Figures E5-5 through E5-7 [point source particulate phase and surface sediment longitudinal profiles]). Differences in chemical concentrations in the traps among the quarters (for TPAH [17]) and more general differences among locations are likely due to the deposition of different combinations of

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[250] Gas ebullition-facilitated transport of contaminants from sediments is another potential source that could have contributed to chemicals in the sediment traps in some portions of the Study Area. This process cannot be a dominant determinant of concentrations in areas where the sediment trap data are lower than the nearby surface sediment concentrations. In other areas, where sediment trap and surface sediment concentrations are similar, contributions from this mechanism would have involved the transport of NAPL/contaminants associated with gas bubbles to the water surface as sheens, where they would need to have associated with solids and then resettled into the sediment traps and onto the surface of the sediment bed (see Section 6.4.7). The extent to which this mechanism would have influenced the sediment trap data depends on the mass fluxes of chemicals from gas ebullition, as well as solids settling fluxes. As such, the importance of gas ebullition as a contributor to chemical loads from the sediment is being evaluated based on FS field studies and the CFT modeling, which will be documented the FS Report.

local sediment and solids from the various sources (e.g., East River, point sources, solids transported between reaches) as well as influences from other potential sources of contaminants and transport processes, such as shoreline seeps (where present), lateral groundwater discharge, and gas ebullition.

Temporal differences in sediment trap TPAH (17) concentrations might be explained in part by differences in precipitation or vessel propwash, as follows:

- With respect to precipitation, total rainfall during Q2 was approximately 13 inches, as compared to 9 inches in both Q1 and Q3. Additionally, total daily rainfall of greater than 1 inch occurred on 6 days in Q2, as compared to only 2 days in Q1 and 1 day in Q3. The higher rainfall (total and intensity) in Q2 likely resulted in greater point source discharges. The effects of higher rainfall would primarily affect the volume of water and loading of solids and chemicals entering the Study Area. Differences in rainfall total and intensity could also result in concentration differences of solids (e.g., as measured in sediment traps). Specifically, differences in rainfall total and intensity can result in different parts of the stormwater basins contributing solids in differing proportions, different types/sources of solids being transported through the outfalls, and differences in the ratio of stormwater to wastewater in the CSOs.
- With respect to propwash, vessel traffic data for the 3-year period from 2009 to 2011 suggest somewhat higher overall traffic (total ship days) in the fall (Q2) and winter (Q3) as compared to the summer (Q1), which may also explain some of the observed variability.

However, definitive conclusions regarding the source of observed variability in sediment trap TPAH (17) concentrations cannot be reached based on the available data. The elevated TPCB concentrations observed in the Q2 samples from one location each in Dutch Kills and Maspeth Creek cannot be attributed to any single potential source and/or mechanism either.

Overall, the sediment trap samples are interpreted to reflect a combination of solids sources and processes (e.g., East River, point sources, and propwash) that varies spatially and temporally.

Overall, the data and evaluations described in this subsection indicate the particulate phase processes of deposition of chemicals associated with sources of external solids (i.e., from the East River and point sources), as well as localized resuspension and redeposition, are important sediment/water exchange processes for evaluating CFT in the Study Area. Fate and transport of particulate phase chemicals will be further evaluated using the CFT model being developed to support the FS (based on predictions from the sediment transport model; see Section 5 of Appendix G).

6.4.4 Surface Sediment Chemical Fate and Transport Processes

This subsection describes the processes that affect chemical concentrations in surface sediment. Processes that act as chemical sources to the surface sediment (e.g., deposition or porewater advection) and those that act as sinks, resulting in a reduction of mass in the surface sediment (e.g., propwash or sediment/water exchange), are discussed. Mixing within the surface sediment layer (i.e., upper 15 cm [6 inches]) is also described. Finally, the ways in which these processes interact to result in changes in concentration over time are summarized. Although there are portions of the Study Area that are subject to episodic erosion/deposition (e.g., sediment mounds that experience localized erosion during point source discharges and areas eroded by vessel propwash; see Section 6.3.4), the Study Area is predominately depositional in nature. As discussed in the remainder of this subsection, the net depositional nature of the Study Area and lower concentrations of more recent depositing solids combine to act as a natural recovery process. Understanding these processes is critical for the CSM development (see Section 8), as well as for evaluation of future changes in surface sediment concentration associated with remedial alternatives in the FS.

6.4.4.1 Definition of the Surface Sediment Layer

The surface sediment layer is defined operationally by the thickness of sediment sampled in the Phase 1 and Phase 2 field programs (i.e., 15 cm [6 inches]), as discussed in Section 4.2.1. This thickness generally represents the biologically active layer that is most relevant to evaluating risk and exposure, although it is likely an overestimate based on literature (USEPA 2015a) and the disturbed nature of the benthic community (see Section 7; see also Section 8.3.2 of Appendix I). The high-resolution core data (presented in Section 4.3.4.2 and discussed further later on in this subsection) provide additional insights regarding the depth of mixing within this layer, based on evaluations of chemical concentration gradients (i.e., mixing is likely limited to be within the top 15 cm [6 inches] or less and does not extend down into the subsurface sediment). This operationally defined thickness of surface sediment is coincident with the biologically active zone (although likely conservatively thick), but is not necessarily meant to represent recent deposition, which varies spatially within the Study Area, as discussed in Section 6.3.4.

6.4.4.2 Sources of Chemicals to the Surface Sediment

The sources of chemicals to the surface sediment of the Study Area can be grouped into two categories: sources that are specific to the Study Area and sources that are regional in nature. Insights gained from hydrodynamics and sediment transport data and modeling have helped differentiate between Study Area-specific and regional sources of chemicals to the surface sediment. As discussed in Section 6.3.2, sediment is transported into Newtown Creek from the East River during each flood tide. A portion of the East River sediment load brought in with the tide is deposited due to the lower energy within the Study Area, with the highest deposition rates near the creek mouth and decreasing rates upstream. NSRs based on multiple empirical LOEs (see Figure 6-5) range from 4 to 5 cm/year in CM 0 to 0.5 to approximately 0.5 to 2 cm/year in CM 0.5 to 2. The net sedimentation in CM 0 – 2 is composed of a large fraction of solids originating from the East River, based on sediment transport modeling (see Figure 6-6). Thus, recently deposited sediments from the East River represent a significant fraction of the mass of solids contained within the 15-cm-thick (6-inch-thick) surface sediment layer. In the upper tributaries, deposition of solids is dominated by point sources (i.e., CSOs and MS4s), as shown in Figure 6-6. These represent sources of solids and chemicals to the surface sediment in these reaches. In CM 2+, solids depositing on the surface sediment (a source of chemicals to surface sediment) originate as a mix of solids from the East River and the point sources. Chemicals sorbed to depositing solids have an important impact on chemical concentrations in the surface sediment of the Study Area.[251]

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[251] In addition to sources of depositing solids, other external sources of chemicals to the surface water (such as lateral groundwater discharge and shoreline seeps; see Section 6.1 and Figure 6-1) can also contribute contaminants to the surface sediment indirectly, through sorption onto depositing solids.

Advection of chemicals present in porewater from the upper zone of the subsurface sediment (operationally, starting at depths of 15 cm [6 inches] below the mudline) represents an additional Study Area-specific source of chemicals to the surface sediment—this source is represented by rates of groundwater seepage (see Section 5.2.1) and measured concentrations within the porewater from a depth just below the surface sediment (i.e., 15 to 30 cm [6 to 12 inches]; see Section 4.8.2). The process of advective transport of dissolved phase chemicals in porewater, which is driven by groundwater flow, is slowed by retardation effects due to contaminant sorption to sediment (e.g., Freeze and Cherry 1979). This dissolved phase advection process acts in a direction opposite to burial from deposition of new sediment at the surface, which effectively moves sediment further down from the sediment/water interface (see Section 6.4.4.3). For chemicals with strong sorption, the effects from this downward deposition of particulate phase mass typically outweigh the effects from upward advection of dissolved phase mass in porewater. As such, porewater advection is relatively more significant as a source of chemicals to the surface sediment in areas with lower relative NSRs and higher seepage rates, primarily for less sorptive contaminants (e.g., LPAHs). Advection within subsurface sediment porewater is discussed further in Section 6.4.5, and will be further quantified with the CFT model, including an evaluation of how the relative balance between the advection process and deposition differs among Study Area reaches.

As discussed previously, porewater concentrations and porewater advection can be impacted by tidal exchanges. As part of this process, surface water can enter the surficial sediments during periods of higher tides, as evidenced by porewater seepage reversals observed in the USGS seepage studies (see Attachments Bi-B9 and Bii-B1 of Appendices Bi and Bii, respectively). Although there are no direct contaminant measures associated with tidal exchange, the lines of evidence presented in Section 6.4.3.1.2 indicate that this process is not a primary driver of shallow porewater concentrations (both 0- to 15-cm and 15- to 30-cm depths) or contaminant fate and transport in surficial sediments.

Other potential sources of contaminants to the surface sediment include migration of NAPL and/or sheen-bearing material[252] associated with advection and/or gas ebullition. NAPL

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[252] The extent to which sheens are a potential source of contaminants to surface sediment is accounted for by subsurface sediment and porewater chemical data; samples were collected over intervals where sheens were reported, and the concentrations quantify the mass of chemicals in the sheen-containing intervals.

transport processes are discussed in Section 6.4.7, and further detail on gas ebullition is presented in Section 2.3 of Appendix D.

Quantitative load estimates of the source contributions to the surface sediment (e.g., through sediment deposition and porewater advection from the shallow subsurface sediment) are developed in Section 6.5.

6.4.4.3 Losses of Chemicals from the Surface Sediment

The mass of chemicals contained within the surface sediment layer is reduced as a result of several processes. As discussed in Section 6.4.3, fluxes from surface sediment to the overlying water column, including both particulate phase (resuspension primarily due to vessel propwash) and dissolved phase (porewater advection and diffusive [including tidal] exchange) mechanisms, are processes by which some mass of chemical is transferred out of the surface sediment layer. Degradation also represents a potential loss process for some chemicals in the surface sediment; discussion of degradation is provided in Section 6.4.6.3. Finally, some chemical mass can effectively exit the surface layer through the process of burial. As a new layer of solids with lower contaminant concentrations deposits onto the sediment bed, the previously existing surface layer becomes buried, with a portion effectively becoming subsurface sediment. This build-up of sediment results in a net “downward movement” of chemical mass from the surface layer to the underlying subsurface sediment.

Quantitative estimates of the chemical mass loads associated with several of these processes are developed in Section 6.5 to allow for their relative magnitudes to be compared, as well as for comparison to the sources (as described in the previous subsection).

6.4.4.4 Physical Mixing in the Surface Sediment

Surface sediment concentrations reflect mixing of depositing solids and current surface sediment, which includes sediments affected by historical loads. In a net depositional environment such as the Study Area (see Section 6.3.4), mixing within the surface layer controls the rate at which surface sediment concentrations change over time as a result of differences in contaminant concentrations between depositing sediment and the existing surface layer sediment. The surface layer sediment concentration is reduced by deposition of solids with lower concentrations and increased by the deposition of higher concentration solids. The rate of concentration change is inversely proportional to the depth of mixing. A thinner mixing layer is associated with more rapid changes in surface concentration by deposition than a thicker mixing layer. The rate of mixing also affects the rate of concentration change. For a given deposition rate, faster mixing generally slows the rate of change within the surface layer, because more of the existing sediment is blended with the new sediment (assuming the new sediment contains different concentrations). Thus, the surface layer represents a blending of newly deposited sediment with previously deposited sediment.

Mixing is a top-down process limited to a relatively shallow depth within the surface sediment (i.e., generally 15 cm [6 inches] or less). Mixing does not move chemical or sediment mass currently present in the subsurface sediment upward into the surface layer (because the depth of mixing does not extend down into the subsurface sediment). However, some fraction of the chemical mass present in the surface layer may have been deposited long ago (and remains there because of mixing). When considering the long history of chemical loads and deposition within the Study Area (as discussed in Section 3.2), this mixing process, when occurring at a significant rate, has resulted in a continual blending of newly deposited sediment with the surface sediment, such that historically elevated concentrations associated with legacy contamination are still reflected in the sediment surface today in several reaches of the Study Area. The extent to which remnants of historical deposition affect concentrations within the surface layer depends on the relative rates of mixing and net sedimentation; legacy contamination affects current surface sediment concentrations most prominently in areas where NSRs are relatively lower and subsurface concentrations are substantially higher than those in surface sediment, such as CM 2+. The relative importance of these processes will be evaluated quantitatively with the CFT model.

The surface layer of the sediment bed in the Study Area may be mixed due to bioturbation and physical processes, as follows:

- The extent of mixing by bioturbation depends on the abundance and types of benthic invertebrates. Benthic community data collected from Phase 1 and Phase 2 surveys and species abundance information presented in Section 8.3.2 of Appendix I (see Table 8-4 of Appendix I) were reviewed to evaluate the potential for bioturbation in the Study Area. These data show that benthic organisms that feed below the sediment surface, which are responsible for mixing sediment via bioturbation, were found to be present in all reaches of the Study Area. Such organisms were observed during multiple surveys, with some seasonal and year to year variation observed. Based on the dominant species observed throughout the Study Area during the benthic community surveys, and according to literature information of the observed species (e.g., Forbes and Lopez 1990), the types of organisms found are generally active within the top 15 cm of the sediment bed.

- With respect to physical mixing, turbulence caused by ship and barge traffic likely represents the dominant process. As discussed in Section 6.3.3, propwash resuspension and subsequent redeposition, as evidenced by sediment trap data discussed in Section 6.4.3.3 and based on the modeling presented in Sections 5.3 and 5.5.3 of Appendix G, serves as a mixing process within the Study Area. Propwash from ships may occur within the navigation channel, from the creek mouth at the East River through the lower portions of English Kills and within Whale Creek (see Section 5.3.2 of Appendix G). The heaviest ship traffic occurs in CM 0 – 1, with decreasing ship traffic in the areas upstream of CM 1.

The high-resolution sediment core profiles (presented in Section 4.3.4.2; see Figures 4-92a through 4-92d) provide an indication of the depth and extent of mixing (or lack thereof) within the surface sediment. Further evaluation of these data is provided in Section 6.4.4.5.

Overall, sediment mixing, along with other previously described fate and transport processes (e.g., deposition), is an important determinant of surface sediment concentrations and concentration changes over time, which are discussed in Section 6.4.4.5. The mixing process will be evaluated further as part of the CFT modeling effort, including consideration of near-surface gradients in radionuclide (lead-210 [Pb-210]) data.[253]

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[253] Pb-210 profiles generally provide a better means to evaluate surface mixing than Cs-137, because Pb-210 is deposited into sediment at a constant rate, whereas Cs-137 provides a known peak in activity at a specified time horizon (i.e., atmospheric nuclear testing in 1963) that occurred prior to what is considered current conditions for both NSRs (see Section 6.3.4) and concentrations of depositing solids. A constant rate is better, since resulting profiles are less sensitive to single event disruptions or variations over time. Also, Pb-210 decays at a faster rate than Cs-137, which results in declines in activity with depth in the subsurface sediment where there are no other ongoing inputs.

6.4.4.5 Changes in Surface Sediment Concentration over Time

Fate and transport of surface sediment also has a temporal component. Contaminant concentrations in surface sediment may increase or decrease over time, depending on the interplay between sources and loss mechanisms. Natural recovery refers to the process by which chemical concentrations in surface sediment decline over time. It is a key consideration in all contaminated sediment investigations (USEPA 2005a); the presence and rates of recovery can vary dramatically between sites.

In Newtown Creek, surface sediment concentrations are declining over time; natural recovery is ongoing throughout the entire Study Area. This conclusion is demonstrated by the observation that for TPAH (17), TPCB, and Cu, the concentration in the first subsurface sediment core segment (usually the depth interval of approximately 15 to 60 cm [6 to 24 inches]) is greater than the concentration in the surface segment (0 to 15 cm [0 to 6 inches] depth interval; see Section 4.3.4.1) in the majority (more than 80%) of the sediment cores.[254] Additional evidence includes the decline in concentrations often observed in the near-surface layers of the high-resolution cores (see Figures 4-92a through 4-92d) and the vertical patterns observed in the majority of cores (see Figures 4-67a through 4-67j, 4-79a through 4-79j, and 4-87a through 4-87j).

Burial by solids with lower chemical concentrations is the primary mechanism by which surface sediment concentrations are declining over time. NSRs impact the rate of burial and decline (higher NSRs will result in faster recovery rates; variations in NSRs by reach and over time are shown in Figure 6-5), whereas other fate and transport processes moderate the rate of decline, including porewater advection, gas ebullition-facilitated transport (where present), and physical mixing. However, although these mechanisms affect the patterns in concentration observed in the top several cm of the bed, they do not change the conclusion that, overall, concentrations in surface sediment are declining.

The interplay between sedimentation and mixing processes in the top 10 to 30 cm (4 to 12 inches) of the sediment bed is illustrated by the high-resolution sediment core profiles (presented in Section 4.3.4.2; see Figures 4-92a through 4-92d). Concentrations of

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[254] Core data for TPAH (34), C19-C36, 2,3,7,8-TCDD, and Pb showed generally similar patterns (see Section 4.3.4.1).

TPAH (17), TPCB, and Cu increase below approximately 30 cm (12 inches) in all but two cores (NC071 and MC005 are the exceptions; see Figures 4-92c and 4-92d), indicating that over the long term, natural recovery due to burial by less contaminated solids is ongoing, in general.[255] Within the top 10 to 30 cm (4 to 12 inches), flat or inconsistent vertical concentration profiles are present in most cores, suggesting varying degrees of mixing. Vertical profiles in cores from CM 1 – 2 exhibit increasing concentration gradients within a few cm of the surface (see Figures 4-92b and 4-92c), providing the strongest evidence of deposition with minimal mixing in the reach.

Vertical patterns sometimes differ among chemicals at individual core locations. In some cases, this difference may be due to sampling or laboratory variability, but in other cases it may reflect chemical-specific differences in the relative concentrations of depositing solids and surface sediment. Near-surface concentration gradients are caused by deposition of solids with considerably lower (or higher) chemical concentrations than present in the sediment bed; gradients are more limited if depositing solids have concentrations that are similar to those present in the sediment bed, or if mixing is rapid enough to eliminate the gradient. Thus, the patterns may differ among the chemicals, depending on the extent to which concentrations are elevated above reference area concentrations. For example, in core NC154 (collected at CM 0.16; see Figure 4-92a), TPAH (17) concentrations exhibit no consistent trend from the surface to a depth of approximately 30 cm (12 inches), whereas TPCB and Cu concentrations increase more consistently from the topmost segment downward, suggesting limited depth of mixing. Thus, at this location, sediment is likely mixed only within the top few cm, and the small differences and lack of a pattern in TPAH (17) concentrations are likely due to the surface sediment concentrations being similar to depositing solids that are at reference area concentrations. Indeed, TPAH (17) concentrations in the top 30-cm (12-inch) range from approximately 20 to 30 mg/kg, and the concentration at a depth of 60 cm (24 inches) is 33 mg/kg, all of which are well within the range of reference area data (see Figure 4-17). In contrast, the high-resolution core located in Whale Creek (WC012; see Figure 92b) shows consistent vertical patterns for TPAH (17), TPCB, and Cu, characterized by a lack of gradient over the upper 10 cm (4 inches), with

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[255] The other contaminants considered in the nature and extent evaluation that were analyzed for in the high-resolution cores (i.e., TPAH [34] and Pb) showed similar behavior. (C19-C36 and 2,3,7,8-TCDD were not analyzed in high-resolution core samples.)

more prominent concentration increases at the 20-cm (8-inch) depth. This pattern is suggestive of a 10-cm (4-inch) mixed depth.

When considered as a whole, the high-resolution core data show evidence of burial over longer timescales in the Study Area, but indicate varying degrees of mixing (it should be noted that not every sediment core can be interpreted as clearly as the examples described in the previous two paragraphs, due to local variability).

Overall, the roles of net sedimentation, sources, loss processes, and mixing have likely combined to produce reductions in surface sediment concentration over time (as compared to historical concentrations) throughout much of the Study Area, as evidenced by the sediment core data. The relative magnitudes of the chemical sources and sinks as compared to the mass contained within the surface layer are compared in Section 6.5, to evaluate spatial differences across the Study Area and to infer the rates at which the surface sediment mass is "turned over." These rates of change (i.e., natural recovery rates) and the mechanisms controlling them will be further evaluated and quantified in the FS (using modeling and other LOEs, such as the Pb-210 data [see Attachment G-G of Appendix G]), as they will affect the evaluation of remedial alternatives.

6.4.5 Subsurface Sediment Chemical Fate and Transport Processes

This subsection describes the processes that affect chemical concentrations in subsurface sediment. Similar to the discussion in Section 6.4.4, processes that act as chemical sources to the subsurface sediment (e.g., transport by burial from the overlying surface sediment and groundwater flow from the underlying native material) and loss processes (e.g., porewater flux to the surface sediment) are discussed. Additionally, the role of sorption/desorption as an attenuation mechanism for groundwater flux within the subsurface sediment is discussed.

6.4.5.1 Sources of Chemicals to the Subsurface Sediment

Chemical mass can be added to the subsurface sediment from the overlying surface sediment layer by burial, as discussed in Section 6.4.4.3. Chemicals can also enter the subsurface sediment from underlying native materials with groundwater flow, which under current conditions, is quantified through the groundwater load calculations presented in

Section 5.2.2 (see Table 5-16) and is compared to the chemical mass present in the subsurface sediment in a mass balance evaluation in Section 6.5.[256] Under current conditions, the mass fluxes of chemicals associated with these two processes are small compared with the total chemical mass that is present in the subsurface sediment, even on the timescale of many decades (see Section 6.5).

The vertical distributions of chemicals in subsurface sediment reflect the long and dynamic history of chemical loads, transport, and deposition within the Study Area. Concentrations of TPAH (17), TPCB, and Cu generally increase with increasing subsurface sediment depth (see Section 4.3.3 and Figures 4-66 through 4-87j) throughout the Study Area. This indicates that historical chemical loads were larger than current loads. Over time, as chemical concentrations in surface water and on depositing solids decreased, concentrations within the surface layer also decreased, as new sediment mixed with the existing sediment and the sediment bed grew in thickness. This process resulted in the general pattern of natural recovery in the observed core profiles, as discussed in Section 6.4.4.

In some reaches of the Study Area, chemical concentrations remain elevated to the bottom of the subsurface sediment at the interface with native material (e.g., TPAH [17] in English Kills; see Figure 4-66). In other reaches, chemical concentrations peak within the subsurface sediment and then decrease with depth toward the native material interface (e.g., TPCB in CM 1 – 2; see Figure 4-78). Even in the presence of such generalized trends, there is variability among cores within a given reach (see Figures 4-67a through 4-67j, 4-79a through 4-79j, and 4-87a through 4-87j).

The vertical patterns in the subsurface sediment concentrations also differ among chemicals. In general, the peak concentration of TPAH (17) is found deeper within the sediment bed than that of TPCB and Cu. For example, in CM 0 – 1, the highest arithmetic average TPAH (17) concentration is found at a depth of 200 to 300 cm (approximately 7 to 10 feet), and the highest arithmetic average TPCB concentrations are found at a depth of 100 to 200 cm (approximately 3 to 7 feet) (see Section 4.3.3 and Tables 4-27 and 4-30). Similar

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[256] Contaminants in groundwater that discharges to the Study Area include contributions from any number of potential upland sources, adding complexity to the understanding of historical and ongoing contamination to the Study Area.

patterns for TPAH (17) and TPCB are observed in other portions of the Study Area. At a given location, deeper sediment reflects older deposition, and these patterns are generally consistent with what is known about historical loads. However, at any given location, the long and dynamic history of loading, transport, and deposition prevents the definitive linkage of observed chemical concentrations to proximate upland sites or sources (including point sources). Much of the historical PAH loads likely occurred in the nineteenth century and the beginning of the twentieth century (see Sections 3.2.6.15 and 3.2.8.1), while peak PCB loads likely occurred closer to the mid-twentieth century, based on the general history of overall PCB use and its regulation within the United States (Versar 1979). As discussed in Section 3.2.6, there were several historical sources of PCBs to the Study Area. The peak depths for Cu (see Table 4-32) tend to be closer to those of TPCB or between those of TPAH(17) and TPCB, which suggests a timing of historical loads between those of TPAH (17) and TPCB. Thus, these vertical patterns support the overall conceptual model of the sediment profile, reflecting sequential chemical and sediment loading history to the Study Area, with higher historical loads followed by progressively lower loads in more recent years (e.g., within the last 10 to 20 years, as compared to 50 to 100 years ago).[257]

The chemical loads associated with current sources to the subsurface sediment (i.e., burial from surface sediment and groundwater) are further evaluated in the context of a mass balance in Section 6.5.

6.4.5.2 Losses of Chemicals from the Subsurface Sediment

Reductions of chemical mass from the subsurface sediment occur due to groundwater flow transporting dissolved and DOC-bound chemicals out of that layer. Additionally, degradation processes represent a chemical mass loss process for some chemicals and under certain conditions (see Section 6.4.6). These mass losses are generally minor on short (annual) timescales, because of the following: 1) the mass of chemicals contained in the subsurface sediment is much larger, as it represents a long (decadal) history of deposition; and 2) the sorptive nature of TPAH (17), TPCB, and Cu limits the amount of mass present in

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[257] Interpretation of vertical chemical distributions in sediment at some locations may be complicated by periodic historical dredging that occurred in various portions of the Study Area (but which has generally not occurred since the 1950s or 1970s, as discussed in Section 3.2.4).

the dissolved phase available for transport by advection and degradation (when applicable—i.e., degradation does not occur for Cu; see Section 6.4.6). The impact of dissolved phase transport from subsurface sediment to surface sediment driven by groundwater flow is further limited by retardation effects and the continual burial of sediment from the surface, as discussed in Section 6.4.4.2; this process will be further evaluated and quantified using the CFT model that is being developed to support the FS.

The presence of Category 2/3 NAPL in the subsurface sediment was observed at three discrete areas within the Study Area (see Section 4.6). Mass transport can potentially affect the NAPL through NAPL advection; gas ebullition-facilitated NAPL transport; and dissolution and subsequent dissolved phase transport via diffusion and advection in porewater. These represent additional mechanisms that can potentially transport chemicals from the subsurface sediment to the overlying surface sediment and surface water in these areas. Discussion of these processes, including data to evaluate their likelihood and potential significance, is presented in Section 6.4.7.

6.4.5.3 Sorption and Desorption in the Subsurface Sediment

Unlike surface sediment, which represents a more dynamic environment with respect to the mass of chemicals due to continual deposition, mixing, and burial, the subsurface sediment acts more as a stable repository for chemicals. Mixing does not extend significantly down into the subsurface sediment because the primary mechanisms of mixing—propwash and bioturbation—are top-down processes that generally do not penetrate much beyond the 15-cm-thick (6-inch-thick) surface sediment layer (see Section 6.4.4).[258]

Chemicals that migrate from groundwater within the native material into the lower portion of the subsurface sediment column subsequently partition between the sediment and porewater based on sediment characteristics and chemical properties (i.e., partition coefficients; see Section 6.4.1). Due to the relatively long hydraulic residence time of groundwater in the subsurface sediment bed (e.g., weeks to months, or longer, based on the

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[258] Although gas ebullition could theoretically contribute to mixing, gas ebullition has not been observed to occur everywhere in the Study Area (see Section 6.4.7.5), and forces from gas ebullition that can move sediment particles are much less than those from bioturbation or propwash. Mixing will be evaluated further as part of CFT modeling.

seepage rates presented in Section 5.2.1),[259] it is likely that sorption equilibrium is attained (e.g., Pignatello and Xing 1996). Thus, given the partition coefficients observed in the sediment (see Section 6.4.1.3), chemicals originating in native materials are likely sorbed to a large degree within the subsurface sediment bed. In areas where groundwater flow is upward and as porewater slowly travels up through the subsurface sediment, chemicals continually sorb to and desorb from the sediment phase, reflecting a state of local equilibrium. As part of this process, it is important to recognize that given the high observed partition coefficients for PAHs, PCBs, and Cu, the mass present in the dissolved phases (i.e., porewater, which includes freely dissolved and DOC-bound phases) is almost negligible compared to that in the sorbed phase.[260] As such, chemicals transported to the overlying surface sediment layer by porewater advection largely originate from the porewater of the upper layers of the subsurface sediment. The decrease in dissolved phase concentrations of TPAH (17), TPCB, and Cu in groundwater that migrate from native material to the lower portion of the subsurface sediment, as compared to concentrations just below the surface sediment (i.e., shallow porewater from the 15- to 30-cm [6- to 12-inch] depth interval; see Section 4.8.2), illustrates the large amount of sorption and attenuation in the subsurface sediment.[261] For example, at most of the locations in the Study Area where higher TPAH (17) concentrations are observed in groundwater, the concentrations in the shallow porewater just beneath the surface sediment, which also represent a state of local equilibrium with the sediment (see Section 6.4.3.1.2), are lower by one or more orders of magnitude (see Figures 4-235a and 4-235b). Differences between groundwater concentrations and

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[259] A simple travel time calculation illustrates this process. Groundwater seepage rates range from net negative (downward) to 8 cm/day upward. Based on a typical subsurface sediment porosity value of 0.75, the linear velocity at which porewater travels upward is 11 cm/day or less. At this upper-bound rate, porewater may travel through 10 feet of subsurface sediment (the average thickness) in about 30 days.

[260] For example, at the lowest K_d value of $10^{3.8}$ L/kg (for C3-naphthalenes, see Figure 6-8c and Table 6-2b), the fraction of dissolved phase chemical mass for sediment with a typical porosity of 0.75 and a typical dry bulk density of 0.6 gram per cubic centimeter (g/cm^3) is less than 0.02%. The fraction of dissolved phase mass for other chemicals with much higher K_d values are orders of magnitude less. Thus, sorption contributes to significant retardation of chemicals being transported by advection in the subsurface sediment. Even for the most mobile contaminants, this calculation indicates that dissolved phase transport is slower than the rate of groundwater advection by more than three orders of magnitude.

[261] In addition to sorption, which retards the rate of chemical transport in the subsurface, there are other mechanisms that attenuate dissolved phase concentrations include dispersion, dilution and degradation—which even if occurring at a slow rate, could affect concentrations over the long timescales of transport (see Section 6.4.6).

shallow porewater concentrations are less for TPCB and Cu, but elevated concentrations of these chemicals generally do not occur in groundwater.

The large degree of sorption and attenuation that occurs within the subsurface sediment dissolved phases, as well as the stable chemical repository of that bulk sediment, are key elements for the CSM (see Section 8). Further evaluation of these concepts, based on mass loads associated with the source/sink terms (i.e., burial from the surface sediment layer, groundwater load entering the bottom of the subsurface sediment, and porewater load exiting the top) and calculations of mass inventory of the subsurface sediment column, is provided in Section 6.5. Additionally, the effects of groundwater loadings on subsurface and surface sediment will be further evaluated and quantified using the CFT model that is being developed to support the FS.

6.4.6 Chemical Reactions and Losses

In addition to partitioning (see Section 6.4.1), other physical, chemical, and biological processes can result in transformations of chemicals within sediment and surface water. Examples include changes in speciation associated with oxidation/reduction and precipitation for metals, and reactions that result in a loss of mass from the sediment or surface water for organic chemicals. These loss processes include volatilization (for certain chemicals), abiotic degradation (e.g., photolysis), and biodegradation, with volatilization and photolysis pertaining to surface water only. These changes and processes are discussed in Sections 6.4.6.1 through 6.4.6.3.

6.4.6.1 Metals Speciation and Precipitation

Sorption, oxidation/reduction, precipitation, and complexation reactions[262] can impact metals bioavailability, as well as fate and transport, significantly. As discussed in Section 4.2.5.3, the AVS present in sediment can react with divalent metals, forming insoluble metal complexes.

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[262] **Sorption**—the physical adherence or bonding of dissolved molecules onto the surface of a solid particle, thereby altering the phase of that molecule from dissolved phase to sorbed phase; **oxidation/reduction**—any chemical reaction in which the oxidation number of a molecule, atom, or ion changes by gaining or losing an electron; **chemical precipitation**—the creation of a solid from a solution; **complexation**—a central atom or ion, which is usually metallic, binding with a surrounding array of molecules or ions, in turn known as ligands or complexing agents

In Newtown Creek, sulfide levels are high enough that these reactions are likely significant, because the AVS levels exceeded the sum of SEM in nearly every sample tested (see Section 4.2.5.3). Therefore, Cu and other metals (including Pb) are likely to be found predominantly as insoluble complexes with sulfide in many areas of Newtown Creek. Additionally, the relatively high partition coefficients estimated for Cu from Study Area data (see Section 6.4.1.3) illustrate that mobility of Cu is limited to transport of particulate matter.[263] As discussed in the MAM2 (Anchor QEA 2016c), metals speciation and complexation reactions may be considered further as part of the CFT modeling that is being developed to support the FS. For example, spatially variable partition coefficients could be evaluated (if supported based on the current dataset) as a way to quantify geochemical and speciation differences affecting dissolved metals.

6.4.6.2 Volatilization

Volatilization is a process by which dissolved phase organic chemicals can be transferred out of the water column into the air at the air/water interface, with the rate of transfer being generally lower for compounds with higher molecular weights, such as PCBs and PAHs. Volatilization flux is a function of several physical factors, including water depth, current velocity, temperature, and wind speed, and is directly proportional to chemical-specific properties such as diffusivity in water and the Henry's Law Constant (HLC). Representative chemical-specific values for diffusivity and HLC for PAHs and PCBs, from published literature, are provided in Table 6-5. This table demonstrates that the diffusivity varies by less than a factor of 2 among the individual PAH compounds and PCB homologs and decreases with increasing molecular weight. Likewise, the HLC values generally decrease with increasing molecular weight, indicating less volatilization for these compounds. However, there is greater variation in HLC (up to three orders of magnitude) among the compounds. These HLC values indicate that although volatilization of PAHs and PCBs is generally low (in comparison to other compounds such as VOCs[264]), it is more important for the lower molecular weight compounds within these classes. Although the volatilization loss is small, this is something that will be considered during CFT modeling, in order to allow for

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[263] The calculations to estimate site-specific partition coefficients did not specifically account for metal-sulfide complexes, so they are an approximate means of describing phases of Cu in surface sediment and porewater.

[264] For example, benzene (a VOC) has an HLC value of 560 joules per mole (J/mol), compared to naphthalene (a PAH), with an HLC value of 45 J/mol.

accurate prediction of surface water concentrations and ultimately the evaluation of remedial alternatives in the FS.

6.4.6.3 Degradation

Degradation refers to chemical or biologically mediated reactions by which a chemical is mineralized or transformed to different chemical species. The most common degradation processes evaluated for CFT are biodegradation (several organic chemicals biologically degrade under certain conditions) and photolysis (breakdown of the chemical by exposure to sunlight, which is important for certain chemicals and conditions). These processes are more relevant for PAHs than for PCBs (and are not relevant for Cu).

Site-specific data on degradation rates are difficult to collect. As such, these processes will be evaluated further in the CFT modeling by parameterization based on literature values, as well as calibration and sensitivity analysis of the model, as appropriate.[265] Evaluation of degradation processes is important, because even if they are occurring at a slow rate, they may result in an appreciable reduction in chemical mass over long (decadal) time frames.

6.4.6.3.1 Biodegradation

Biodegradation can be a significant process for certain organic compounds (e.g., some lighter PAHs) found in sediment and porewater in the Study Area. This process involves the metabolic oxidation or reduction of organic compounds carried out predominantly by bacteria. Microbes may either gain chemical energy directly as a result of biodegradation of an organic compound, or indirectly during the process of co-metabolism, which is the concurrent degradation of another substrate with the organic compound. Biodegradation rates depend on the following: chemical structure and concentration; the concentration of bacteria responsible for the biodegradation; the availability of organic matter to serve as food and energy sources for bacterial growth; the availability of essential nutrients (nitrogen and phosphorus) for bacterial growth; and physical and chemical conditions at the site, such as

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[265] Based on preliminary simulation of PAH compounds with the CFT model, some level of degradation (combination of biodegradation and photolysis) in the surface water appears to be occurring in the Study Area (as discussed in Section 6.4.3.1.2). This process will be further evaluated as part of the CFT modeling effort.

temperature and oxygen level.[266] The extent to which the organic compound is bound to particles may also affect the biodegradation rate, as more strongly bound organic compounds may be biologically less available for microbial degradation.

Representative order-of-magnitude biodegradation rates of PAHs in surface water and sediment are provided in Table 6-5 (presented as half-lives), based on a literature review.[267] Fewer reviewed documents reported biodegradation of PCBs (as compared to PAHs) and where reported, rates varied significantly—including characterizations as nondegradable in many cases. Also, PCBs can undergo reductive dechlorination (a reaction in which a chlorine atom is cleaved from a PCB molecule, resulting in a formation of a lower chlorinated PCB congener); however, the Study Area PCB composition data do not show much evidence of this process.[268] Due to the limited and inconsistent rates found in the literature for biodegradation and the lack of substantial site-specific evidence for reductive dechlorination, degradation of PCBs is conservatively considered to be negligible over the timescales relevant for the CFT evaluations of the Study Area.

Under most conditions, anaerobic biodegradation is generally much slower than aerobic biodegradation for PAHs and PCBs (Howard et al. 1991). The representative half-life rates listed in Table 6-5 were obtained from BIOWIN, a model within USEPA's Estimation Program Interface Suite (EPI Suite; USEPA 2012e).[269] This model assumes aerobic conditions, but deeper layers of aquatic sediment are usually anaerobic. To account for the slower rate of biodegradation in the sediment, EPI Suite uses a conversion factor that assumes the rate of biodegradation in sediment is on average one-ninth of that in the water

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[266] Although *aerobic* degradation will consume oxygen and can therefore contribute to an oxygen deficit, the occurrence of this should likely be minimal compared to *anaerobic* degradation. With respect to oxygen deficit, the oxygen consumption associated with more labile organic matter in the sediment, such as that derived from CSOs, is substantially greater than that from aerobic degradation of chemicals.

[267] Aronson et al. (1999), USEPA (2012e; BIOWIN), USEPA (2015b), and Howard et al. (1991)

[268] CBP (see Section 4.2.5.2) is consistently greater than 4 in surface sediment (see Figure 4-47), indicating that widespread, extensive dechlorination has not occurred in Newtown Creek (an inspection of subsurface data shows similar CBP values). The values of CBP in sediments lie in the range of Aroclors 1248, 1254, and 1260, which are common in environmental samples, also suggesting that dechlorination, if it has occurred, has been a minor factor (see Figure 4-47). This can be contrasted with other sites, such as the Upper Hudson River, which show a strong shift to the lower homologs (e.g., Brown et al. 1987).

[269] From the BIOWIN user guide: "Biodegradability estimates are based upon fragment constants that were developed using multiple linear or non-linear regression analyses."

column (which is assumed to be aerobic). For PAHs, the half-lives reported in the EPI Suite generally range between 20 and 100 days for aerobic degradation (water) and 170 to 950 days for anaerobic degradation (sediment). These rates generally agree with those identified from a review of several published sources (see Footnote 267).

However, it is important to note that literature-reported degradation rates are quite variable, including some studies showing little to no degradation for certain PAHs. This reinforces the understanding that "biodegradation is determined by the physical, chemical and biological parameters of the environment in which the contaminant is found, as well as the physical/ chemical properties of the contaminant itself" (Aronson and Howard 1997).

Given the chemical mass present in the subsurface sediment of the Study Area (see Section 6.5), anaerobic degradation of PAHs, even if occurring at a slow rate, may result in a significant loss of mass over long (decadal) timescales. Thus, this process will be considered further as part of the CFT modeling effort in the FS (and given the large degree of variability in the literature, sensitivity analysis will likely be used in these evaluations).

6.4.6.3.2 Photolysis

Photolysis reactions occur in response to absorption of solar energy (ultraviolet [UV] band) and can occur either directly or indirectly. Direct photolysis occurs when molecular bonds are broken by electromagnetic radiation (particularly high-energy UV radiation). Indirect photolytic reactions occur when reactive species (such as a hydroxyl radical or singlet oxygen) are formed, which subsequently react with organic molecules. The degree to which photolysis occurs is affected by the depth and light extinction in the water column (e.g., due to turbidity and other factors), and by the intensity and angle of incidence of light (Chapra 1997). Photolysis in the water column can occur for PAHs and to a lesser extent for PCBs. As with the other reaction processes discussed in this section, photolysis will be considered further, as appropriate, in the upcoming CFT modeling.

6.4.7 NAPL Fate and Transport Processes

As discussed in Section 4.6 (see also Section 6.3 of Appendix C), NAPL has been observed primarily as discrete droplets at apparent residual saturation in sediment and native material in the Study Area (i.e., Category 1B).

For the purposes of this RI, NAPL pore fluid saturation (hereafter referred to as NAPL saturation) is defined as the fraction of total sediment or native material pore volume occupied by NAPL (Cohen and Mercer 1993). The remainder of the sediment or native material pore volume is generally filled with water; gas associated with ebullition may also occupy pore spaces in sediment. Residual saturation is the condition in which NAPL saturation is sufficiently low such that the NAPL consists of discrete blebs trapped by capillary forces, and is immobile. This classification is specific to the ability of the NAPL to advect (i.e., flow) as a nonaqueous fluid phase.[270] The interpretation that NAPL blebs represent residual, immobile NAPL is based on the observation that in core samples, the blebs are present as small, discrete droplets; this matches the description of residual NAPL in the literature (Schwille 1988; Cohen and Mercer 1993; Pankow and Cherry 1996; API 2003; ITRC 2004; Sale et al. 2008; ITRC 2009; Kueper and Davies 2009). This interpretation is supported by the results of the FS Part 1 NAPL mobility testing of sediment and native material in CM 0 – 2 (see Section 6.4.7.2; see also Section 4.4 of Appendix C). Observations of qualitatively higher NAPL saturations within the subsurface sediment and underlying native material were limited to three localized areas: in the main stem near CM 1.7, in the Turning Basin, and in lower English Kills. Potential NAPL migration mechanisms and properties that influence NAPL fate and transport are discussed in this subsection and will be used to identify and screen technologies and remedial alternatives for further consideration in the FS.

Although NAPL in sediment is the focus of this subsection, NAPL in sediment is not the only potential source of NAPL to surface water. The NAPL migration mechanisms described in

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[270] Although immobile NAPL will not flow as a separate phase material, it can migrate via gas ebullition. In areas where gas ebullition is occurring, it is possible for immobile NAPL to become entrained in a gas bubble and travel upward. In addition, immobile NAPL can be temporarily disturbed during activities such as anchoring, dredging, and bulkhead repair, or during sediment erosion events caused by vessel propwash or high current velocities (e.g., near outfalls during precipitation events), and subsequently may travel to the surface of the water as a result of such activities.

the Sections 6.4.7.1 through 6.4.7.7 focus on the potential for NAPL and NAPL constituents in Study Area sediment to migrate to surface water. However, potential ongoing NAPL sources (and associated sheens on Study Area surface water) also include stormwater discharges (e.g., MS4 and private discharges), CSO discharges, shoreline seeps, parking lot and roof drain runoff, illegal dumping, spills, and releases from boats and barges operating on Newtown Creek (as well as by transfer from sediment via gas ebullition). Some fraction of the NAPL and NAPL constituents discharged to Study Area surface water from these sources is deposited in the sediment bed.

6.4.7.1 NAPL Migration Mechanisms

Potential mechanisms for the migration of NAPL and NAPL constituents include the following:

- Migration of NAPL as an immiscible fluid flowing through permeable sediment, in response to hydraulic gradients and/or density-driven forces (i.e., NAPL advection)
- NAPL dissolution and subsequent dissolved phase advection of NAPL constituents in groundwater, porewater, and surface water
- NAPL dissolution and subsequent diffusive flux of dissolved phase NAPL constituents within the sediment bed and between surficial sediment porewater and surface water via dissolved phase exchange
- NAPL migration in sediment and surface water associated with gas ebullition
- NAPL and sheen movement on surface water due to wind and currents
    - NAPL may also enter surface water from migration of potentially mobile NAPL, from MS4 and private storm drains, CSOs, shoreline seeps, parking lot and roof drain runoff, illegal dumping, spills, and releases to surface water from vessels. If the NAPL reaches the surface of the water, the NAPL will typically disperse and transform into a thin sheen on the surface of the water.
- NAPL transport and deposition to the sediment bed in the form of oil-particle aggregates (OPAs)

Potential NAPL and NAPL constituent migration mechanisms are described in more detail in the following subsections.

6.4.7.2 NAPL Advection in Sediment and Native Material

At NAPL saturations above the residual saturation, NAPL is continuously connected between pore spaces and has the potential to flow, or advect, if the driving forces acting on the NAPL (i.e., hydraulic and gravitational) are sufficient to overcome the capillary forces that resist NAPL movement.

The ability of NAPL to flow as a separate phase in sediment and native material depends on many factors, including the following:

- **NAPL distribution in the sediment or native material pore network**. NAPL must be interconnected within the larger pores to be capable of advection. The degree to which the NAPL phase is continuous or discontinuous depends largely on the NAPL saturation, which is the percent of the total pore space that is filled with NAPL.
  - Disconnected NAPL, such as separate droplets of NAPL (i.e., blebs), has insufficient NAPL saturation to flow as a continuous fluid. In this case, the NAPL is immobile and is described as an immobile, residual phase NAPL (ITRC 2015), as discussed previously in this section. Residual saturation depends on several factors, including NAPL fluid properties, sediment pore structure, the maximum historical NAPL saturation within the porous medium, and the NAPL emplacement mechanism.
  - If NAPL is interconnected within the pore spaces of the sediment and the NAPL saturation is higher than the residual saturation, then the NAPL is potentially mobile (ITRC 2015).
  - For NAPL to be mobile (i.e., move via advection), the NAPL saturation must be greater than the residual saturation, and its capillary pressure must exceed the pore entry pressure of the surrounding porous media. The capillary pressure is the NAPL pressure minus the water pressure. The pore entry pressure is the capillary pressure that must be overcome for a nonwetting NAPL to enter the largest pores (which offer the least capillary pressure resistance) in a water-saturated medium (Cohen and Mercer 1993).
- **NAPL physical properties**. This factor refers to NAPL density, viscosity, and interfacial tension with water. NAPL physical properties affect the ability of NAPL to flow, including the conditions under which flow may occur and the direction and

distance of flow. For example, as the viscosity of NAPL increases, the potential for NAPL to flow decreases.

- **Driving forces acting on the NAPL**. Multiple driving forces affect NAPL mobility: the NAPL specific gravity (if appropriate), the hydraulic gradient, and the NAPL gradient (i.e., the hydraulic head associated with differences in the vertical thickness of the NAPL layer between two points).
- **Sediment characteristics**. Specifically, the pore size and wettability of the material the NAPL is contained within affect NAPL mobility. Pore size is a function of grain size distribution and porosity; pore size is inversely proportional to the pore entry pressure (i.e., the larger the pore size, the lower the pore entry pressure). Wettability is the tendency of a liquid to preferentially coat solid surfaces.
- **NAPL emplacement**. This factor refers to the mechanism by which NAPL came to be present in the sediment or native material: subsurface NAPL advection, or NAPL deposition with sediment, or a combination of both mechanisms. These two processes result in differences in how NAPL is distributed in the sediment or native material pore network, and the degree to which the NAPL phase is continuous or discontinuous. Emplacement by subsurface advection requires a continuous NAPL phase. After emplacement by advection, the NAPL may or may not become discontinuous and immobile. Emplacement by deposition along with sediment (i.e., deposition of OPAs) results in tiny droplets of oil that are surrounded by solid particulates. In this case, the NAPL droplets are typically discrete, disconnected, and immobile within the sediment (Johnson et al. 2018).

All these factors affect the ability of the sediment and native material to transmit NAPL.

As part of the NAPL evaluation, NAPL observations were qualitatively categorized based on the relative magnitude of NAPL observed in the samples, as discussed in Section 4.6.1.3 (see also Section 3.4 of Appendix C). Samples that contained no visible NAPL (negative or sheen shake test results) were characterized as Category 1A samples. Samples that contained visible, discrete NAPL (i.e., blebs observed in shake tests) were characterized as Category 1B samples; these samples are interpreted to contain only residual NAPL. The Phase 2 FSAP Volume 2 and FS FSAP defines a visual observation of blebs in sediment as follows: "Observed discrete sphericals of NAPL but for the most part, the sediment matrix was not visibly contaminated or saturated. Typically, this is residual product" (Anchor QEA 2014a, 2017b).

In limited areas (CM 1.7, the Turning Basin, and lower English Kills, as described in Section 4.6.2 through 4.6.4 [see also Section 5.4 of Appendix C for more detail]), samples in some cores were identified as Category 2/3 samples, which contained apparently higher NAPL saturation values (i.e., shake test layer results) compared to Category 1A and Category 1B samples. The Phase 2 NAPL descriptions provide a basis upon which to qualitatively compare relative estimated NAPL saturations. However, additional information is needed to quantify the actual NAPL saturation and test whether NAPL has the potential to be mobile and, in particular, move upward toward the mudline.

Based on an understanding of the factors that affect NAPL mobility, a staged testing approach was implemented under Part 1 of the FS to evaluate NAPL mobility, as described in Section 4 of Appendix C. The results from each stage of testing are used to determine whether the next stage of testing is required, as described in detail in Section 4.2 of Appendix C. The stages of testing include the following: an initial screening; centrifuge NAPL mobility testing (i.e., Stage 1); flexible wall permeameter mobility testing (i.e., Stage 2); and capillary pressure curve, hydraulic conductivity, bulk density, and porosity characterization (i.e., Stage 3). NAPL saturations are measured as part of Stage 1 and Stage 2 testing.

In the initial screening step, the full length of a frozen core from each of the 11 FS NAPL mobility stations within CM 0 – 2 was photographed under white light and UV light to identify the general depth intervals of visual indicators of potential NAPL (primarily, fluorescence under UV light). These depth intervals were then targeted in the collocated unfrozen cores for further photographic screening and Stage 1 NAPL mobility testing. The purpose of the initial screening was to ensure that the collocated unfrozen core intervals targeted for further photographic screening and Stage 1 NAPL mobility testing represented the conservative, worst-case conditions for that station.

In sediment, some degree of fluorescence was observed in frozen core photography at all 11 of the CM 0 – 2 FS NAPL mobility stations. While the intensity and density of fluorescence varied from location to location, a general pattern of dispersed fluorescence appearing as minute, discrete specks of fluorescence in a non-fluorescing matrix was observed in the sediment. Although the emplacement mechanism for the minute fluorescent specks in the sediment is uncertain, the general pattern of dispersed fluorescence may represent what the U.S. Geological Survey (USGS 2015) described as OPAs (see Section 6.4.7.7). Fluorescence was also observed in native material at 4 of the 11 CM 0 – 2 FS NAPL mobility core stations, in intervals ranging from approximately 1-cm (less than 0.1-foot) up to 1-foot thick. Where fluorescence was observed in native material at CM 0 – 2 FS NAPL mobility core stations, there was no clear pattern in its spatial distribution. NAPL saturation measurements in collocated cores—after Stage 1 NAPL mobility testing—confirmed that NAPL was present in sediment and native material samples where fluorescence was observed, as discussed in Section 4.4.3 of Appendix C.

FS Part 1 Stage 1 mobility testing was performed for a total of 24 sediment samples and 4 native material test samples (collected from Category 1B and Category 2/3 Areas) obtained from unfrozen cores collected from 11 stations located within CM 0 – 2. Stage 1 test samples were selected from specific depths with the highest visible indication of NAPL presence, as identified based on *unfrozen* core photography (i.e., initial screening step). For each Stage 1 target depth interval containing the highest visible indication of NAPL presence based on *frozen* core photography, a section of the collocated *unfrozen* core was processed and shipped to the laboratory. The unfrozen core sections were centered vertically at the same depth as the target depth identified based on frozen core photography. The unfrozen core sections were slabbed and photographed under white light and the UV light (wavelength 365 nanometers) to select the specific sample depth(s) for Stage 1 testing.

In Stage 1, the potential for NAPL mobility was evaluated by centrifuging each Stage 1 mobility test sample at 25 times the acceleration due to gravity (G) for 10 hours. The 25-G centrifuge spin produces a driving force equal to a hydraulic gradient of 25,[271] which is more than 100 times

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[271] As described in Gefell et al. (2019), free gravity drainage of a saturated sediment sample with no additional head applied at the top of the sample produces a hydraulic gradient of 1. Therefore, centrifuging a sample at a force of 10-G would create a hydraulic gradient of 10. These calculations assume a fluid density of 1 g/cm^3; for a NAPL with a density less than or greater than 1 g/cm^3, a 10-G force would create a hydraulic gradient of 10 times the NAPL density. For example, under a force of 10-G, a NAPL with a density of 0.86 g/cm^3 would experience a hydraulic gradient of 8.6, while a NAPL with a density of 1.1 g/cm^3 would experience a hydraulic gradient of 11.

larger than the maximum upward hydraulic gradient measured in the Study Area (0.23),[272] based on multiple months of continuous hydraulic data measured using transducers at long-term monitoring stations as part of the RI groundwater evaluation (see Section 3.4.1 of Appendix F). Thus, compared to Study Area conditions, the Stage 1 mobility test is conservative. Any sample that does not show NAPL mobility under this test condition indicates the NAPL is immobile and cannot advect under the smaller driving forces that exist in situ.

NAPL was not produced during mobility testing for any of the 24 sediment samples and 4 native material samples tested from CM 0 – 2, despite the fact that the Stage 1 samples were selected from depths containing the highest apparent NAPL saturations (see Section 4.4.2 of Appendix C).

Due to the lack of mobility observed in CM 0 – 2 Stage 1 samples, Stage 2 and Stage 3 testing were not performed.

As discussed in Section 4.4.3 of Appendix C, and presented in Table C4-1 of Appendix C, NAPL saturations in Creek Mile 0 – 2 Category 1B Areas and the CM 1.7 Category 2/3 Area are similarly low, and the Stage 1 NAPL mobility testing results indicate that these NAPL saturation values are insufficient to produce NAPL mobility, as follows:

- In Category 1B Areas located in CM 0 – 2, NAPL saturations in sediment samples ranged from 1.0% to 13% of total pore volume (Pv) (17 samples), while saturations in native material samples ranged from 1.8% to 16% Pv (3 samples).
- In the CM 1.7 Category 2/3 Area, NAPL saturations in sediment samples ranged from 1.7% to 6.7% Pv (7 samples), while the saturation in the one native material sample tested was 1.9% Pv.

The NAPL fluid saturation test results indicate that, on the whole, NAPL in sediment and native material in the CM 1.7 Category 2/3 Area is not present in measurably greater amounts than the surrounding Category 1B Areas.

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[272] This hydraulic gradient value is based on multiple months of continuous hydraulic head data measured using transducers at long-term monitoring stations as part of the RI groundwater evaluation (upward hydraulic gradients range from 0 to 0.23; see Section 3.4.1 of Appendix F).

The potential mobility for NAPL in subsurface sediment and native material in the localized Category 2/3 Areas in the Turning Basin and lower English Kills may be an important FS consideration during the development and screening of remedial alternatives. Therefore, NAPL mobility in these areas will be evaluated during Part 2 of the FS. The results of the NAPL mobility testing for the other two Category 2/3 Areas and upstream Category 1B Areas are provided in the FS NAPL DER (Anchor QEA 2022a).

6.4.7.3 NAPL Dissolution and Advection in Groundwater, Porewater, and Surface Water

NAPL constituents may dissolve in groundwater, porewater, or surface water when NAPL is in contact with these media. The NAPL dissolution process is complex, and often rate-limited, due to the chemical composition of the NAPL, diffusion rates within the NAPL and the surrounding water, conditions that affect dissolution of NAPL constituents at the NAPL-water interface, and other factors (including temperature and salinity). The mass of NAPL constituents that can be dissolved in water is a function of the effective solubility (as per Raoult's law) of each of the NAPL constituents. Once present as a dissolved phase, NAPL constituents migrate with the bulk groundwater, porewater, or surface water flow and are subject to the fate and transport processes that are discussed in Sections 6.4.2 through 6.4.6 and represented in the CFT model.

6.4.7.4 NAPL Dissolution and Diffusive Flux Between Surficial Sediment Porewater and Surface Water

As discussed in Section 6.4.3, diffusion is another dissolved phase exchange process. Dissolved phase NAPL constituents present in surficial sediment porewater can diffuse to surface water, due to differences in concentrations between porewater and surface water. Diffusive flux is discussed along with the other dissolved phase fate and transport processes in Section 6.4.3.

6.4.7.5 NAPL Migration Associated with Gas Ebullition

Gas ebullition is the formation of gas bubbles (primarily methane, but may also include carbon dioxide, nitrogen, oxygen, hydrogen sulfide, and other gas constituents) in organic-enriched sediments, due to anaerobic decomposition of the organic material and subsequent migration of the resultant gas bubbles through soft sediment. Through this process, gas ebullition can serve as a potential contaminant migration pathway for NAPL and other contaminants from sediment to surface water.

Organic matter that can generate gas bubbles includes naturally occurring organic matter, organic matter associated with CSO and storm drain discharges (e.g., fecal matter), and other organic contaminants (e.g., NAPL). The rate of microbial activity and gas generation is directly affected by sediment temperature, which changes throughout the year. Once formed, these gas bubbles can migrate upward in sediment, depending on environmental conditions. Factors favorable for gas ebullition include low hydrostatic pressure (e.g., locations with shallow water), cohesive sediment with low strength, low permeability, and high organic content. When gas ebullition occurs in the presence of NAPL, NAPL may be transported with gas bubbles through sediment to the water surface and form sheen blossoms. Sheen blossoms refer to sheens observed to appear on the water surface, associated with breaking gas bubbles.

Qualitative studies of gas ebullition were conducted in August 2015 and September 2016 as part of the Phase 2 investigations. Both studies made observations of the location, frequency, and magnitude of bubble generation and sheen blossoms at the water surface to develop an understanding of conditions where gas ebullition-facilitated NAPL transport is most likely to occur. These FESs were performed during the time of year when gas ebullition is expected to be most active (i.e., August and September), and the surveys are likely a conservative record of observations compared to other times of the year. A quantitative gas ebullition pilot study was conducted in September 2017 to develop and test methodologies for the 2018 to 2019 gas ebullition field program conducted under Part 1 of the FS (data for the 2018 to 2019 gas ebullition field program are not included in the RI Report and are presented in the FS Gas Ebullition DER [Anchor QEA 2022b]).

Additional detail regarding these gas ebullition-related field activities, discussion of the results, and an evaluation of the processes that generate and influence NAPL migration associated with gas ebullition are provided in Appendix D. Key findings of the gas ebullition evaluation are as follows:

- The 2016 FES, which mapped the spatial extent of dynamic sheen and gas bubbles, was conducted near optimal conditions for annual maximum gas ebullition (i.e., summer spring tide, when the sediment temperatures are warmest and the tidal elevations are lowest). Given the conditions under which this study was conducted and the area surveyed, it is considered representative of the near maximum extent of the portions of the Study Area where gas ebullition occurs.
- Based on the RI field ebullition surveys, gas ebullition appears to be limited to areas with water depths shallower than 6 meters (see Section 5.1.2.2 of Appendix D). The 2016 FES included observations in multiple areas with water depths greater than 6 meters, with no observations of sheen blossoms and minimal observation of gas bubbles. More widespread gas bubbles were observed in the tributaries, where the water depths are generally shallower than the deeper water in the main stem.
- Sheen blossom and gas bubble observations during the 2015 and 2016 FESs and the 2017 pilot study differed throughout the Study Area. The differences in observations between the various survey/sampling areas are likely associated with differences in organic material content and quality, as well as the NAPL content in the sediment. Comparison of gas bubble observations and sediment TOC indicates the most significant apparent gas ebullition-related observations were generally in areas with the highest TOC. Other factors, including sediment strength and water depth, may also influence differences in the gas ebullition process.
- The majority of gas ebullition is expected in surficial sediments, where temperatures are warmest and sediment strengths[273] are weakest.
- The occurrence of gas bubbles and sheen blossoms generally increases with lower tidal elevations.
- The daily window for gas ebullition activity is predominantly limited to approximately 2.5 hours prior to and 1.5 hours after low tide, with the maximum gas ebullition rate observed within a 1-hour window around low tide.
- Comparisons of RI surface water chemistry measurements with water depths/tidal elevations and surface water temperature provide lines of evidence regarding the potential impact of gas ebullition on surface water contamination. Correlations were

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[273] In this context, sediment strength refers to a material characteristic of the sediment to resist fracture and formation of bubble migration pathways, which result from stresses induced on the sediment by gas bubble formation and growth.

not found, indicating that neither surface water temperature nor tidal elevation affect surface water chemistry (consistent with the data evaluations presented in Section 4.7.3 and in Appendix D). Shallow surface water samples were collected 1 foot below the water surface, so they likely did not capture sheens from gas ebullition that were in the process of spreading across the water surface before dissolving, partitioning onto solids, and/or potentially degrading. These results, combined with other lines of evidence presented in this report, lead to the conclusion that NAPL/contaminant transport by gas ebullition is not currently one of the largest contaminant migration pathways affecting surface water concentrations.

- Static sheens not related to sheen blossoms were observed in both 2015 and 2016. In particular, significant portions of the Study Area were covered by static sheen after precipitation was observed during the 2016 FES, which appeared in part to be related to point source discharges. On September 19, 2016, sheens were observed by NCG originating from three outfalls: NCB-683, NCQ-637, and BB-609 or BB-610 (because these outfalls are submerged at high tide, static sheen could have entered the outfall[s] due to tidal movement prior to the observations of these sheens originating from these outfalls). Static sheens related to spills were also observed.
- The 2017 pilot study demonstrated the feasibility of measuring NAPL/contaminant and gas flux, using near-bottom flux chambers.
- Visual observations of sheen blossoms and gas bubbles were generally consistent with gas ebullition-facilitated NAPL/contaminant flux and gas flux measurements during the pilot study.

Additional investigation of gas ebullition-facilitated transport of NAPL/contaminants was initiated in July 2018 and continued through July 2019 as part of the FS gas ebullition field program. The program included additional NAPL/contaminant and gas flux measurements (July and October 2018); visual observations of sheens and gas bubbles (2019 visual survey and camera observations); sediment temperature depth profile measurements (July 2018 to July 2019); and measurements/observations during different times of the year to capture the effect of different temperatures on gas ebullition-facilitated transport of NAPL/contaminants. Following this program, the results will be used to extrapolate flux measurements to other times of the year and/or other areas of the Study Area for use in refinement of the RI/FS CSM, development of the CFT model, and to potentially support the development and technology screening of FS alternatives. The results of the 2018 to 2019 FS gas ebullition field program are provided in the FS Gas Ebullition DER (Anchor QEA 2022b).

6.4.7.6 NAPL Movement on Surface Water

NAPL or sheen may reach the water surface in the Study Area from a number of sources, including stormwater discharges (e.g., MS4s), CSO discharges, bank seepage, parking lot and roof drain runoff, illegal dumping, and releases from boats and barges operating on Newtown Creek, as well as by transfer from sediment via gas ebullition.

When NAPL reaches the water surface, it disperses, and depending on the physical properties of the NAPL and the quantity present, the NAPL either breaks into small droplets or transitions to a sheen on the water surface. Surface water NAPL and sheen break down by photodegradation, biodegradation, volatilization, and dissolution. As the NAPL or sheen breaks down, the NAPL or sheen may be transported by wind and surface water flow and can be deposited on the sediment (described in Section 6.4.7.7). The migration process is influenced by surface water currents (summarized in Section 6.2), wind speed and direction, vessel traffic, and the presence of structures (e.g., shoreline bulkheads or containment booms).

6.4.7.7 Oil-Particle Aggregates

Aggregation of oil droplets or sheen residuals suspended in or on surface water, and particles suspended in or on surface water, can form OPAs (USGS 2015). When the particles adhere to the oil droplet, the resulting aggregates become denser than water, causing them to sink within the water column and deposit, becoming part of the sediment bed. Following deposition, the OPAs may be resuspended by erosive flows and gas ebullition that entrains the OPAs and/or surface water currents with velocity sufficient to erode the OPAs (USGS 2015). Surface water flows and associated particulate transport are discussed in Sections 6.2 and 6.3.

6.5 Mass Load and Inventory Comparisons

In this subsection, empirical mass load estimates of chemicals are developed to compare the relative importance of many of the chemical mass transport processes discussed previously in this section with sources to the Study Area, as well as the preliminary chemical mass inventory estimate of the Study Area sediment and water column. The mass load and preliminary chemical mass inventory estimates are derived using the full RI dataset for all field programs and media in which data have been collected. Because the quantitative load estimates are developed to compare the relative importance of various transport processes, they were derived using arithmetic average concentrations. It is recognized that there is uncertainty in these arithmetic averages, but it is not critical to completing the RI to evaluate these potential uncertainties at this time. The mass load processes presented in this section, as well as any uncertainty and variability in these processes, will be further evaluated and refined as part of the CFT modeling effort. Quantitative mass estimates for TPAH (17), TPCB, and Cu are developed and presented for the following terms that are discussed in previous sections of this report and are shown in Figure 6-39:

- Preliminary mass inventory estimates in surface water, surface sediment, and subsurface sediment
- Loads associated with point sources, groundwater, and atmospheric deposition
- Sediment/water exchange processes for both dissolved phase (surface porewater advection and diffusive exchange, which encompasses tidal exchange [where present]) and particulate phase (due to sediment deposition)
- Subsurface/surface sediment mass transfer processes, including burial and porewater advection

There are other processes that may be important, but are not presently quantified (e.g., East River tidal exchange,[274] chemical fluxes from propwash resuspension, degradation processes, shallow lateral groundwater loads, and gas ebullition). These processes will be evaluated further, as appropriate, as part of the CFT modeling effort and the FS.

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[274] In the case of the East River tidal exchange, as discussed in Section 5.3, the water entering the Study Area twice daily during flood tide represents a source of chemicals to the Study Area. The same volume of water exits during ebb tide, and as discussed in Section 5.3, the concentrations measured at the mouth of the creek under ebb and flood tide conditions are very similar (see Tables 5-17, 5-19, and 5-21), due to the large volumetric exchange of water. The net differences between these gross fluxes (i.e., flood versus ebb tide), some of which contribute to chemical deposition flux within the Study Area, cannot be reliably calculated from these data, due to the complex circulation patterns at the mouth of the creek. However, the CFT modeling will allow for quantification of these fluxes.

6.5.1 Methods and Results for Inventory and Load Estimates

Sections 6.5.1.1 through 6.5.1.4 provide a description of the methods used to calculate each of the mass load/inventory terms shown in Figure 6-39. A summary of the resulting estimates by reach and summation for the Study Area is presented in Table 6-6. Discussion of the results is provided in Section 6.5.2.

6.5.1.1 Preliminary Chemical Mass Inventory Estimates

A preliminary estimate of the chemical mass inventory for the surface water and sediment was calculated as follows:

- **Surface water**. The arithmetic average chemical mass present in the surface water of the Study Area was estimated for each reach based on arithmetic average dry and wet weather surface water chemical concentrations (see Tables 4-49, 4-51, and 4-54 [dry weather] and Tables 4-56, 4-58, and 4-60 [wet weather]). The arithmetic average concentrations were multiplied by the arithmetic average volume of water in the reach (based on its plan view surface area and arithmetic average water depth relative to mean sea level, as listed in Table 6-7). The arithmetic average concentration used in the calculation was based on a weighted average of the dry and wet weather data using weighting factors of 0.8 and 0.2, respectively (based on the approximate number of days that each condition typically occurs).
- **Surface sediment**. The preliminary chemical mass inventory for surface sediment (0- to 15-cm [0- to 6-inch] depth) was estimated based on the arithmetic average concentration by reach (see Tables 4-13, 4-16, and 4-18; for TPAH [17], TPCB, and Cu, respectively) multiplied by the 15-cm (6-inch) surface layer thickness, the surface area of the reach (see Table 6-7), and the average dry density of the surface sediment for the reach (see Table 6-8).
- **Subsurface sediment**. The subsurface sediment (15-cm [6-inch] depth to the sediment/native material interface) preliminary chemical mass inventory was estimated using the same general approach as for surface sediment, except for the inclusion of multiple layers, and the sediment thickness. For each of the depth intervals used for concentration averaging (i.e., 15 to 60 cm [0.5 to 2 feet], 60 to 100 cm [2 to approximately 3 feet], 100 to 200 cm [3 to approximately 7 feet]], etc.; see Tables 4-27, 4-30, and 4-32 for TPAH [17], TPCB, and Cu, respectively), the

concentration was multiplied by the interval thickness, the corresponding average dry density (see Table 6-8), and the surface area of sediment within the reach for the given depth interval. Thiessen polygons of the native material depth observations in continuous subsurface sediment cores were used to assign total sediment thicknesses and to spatially specify the surface area of subsurface sediment present in each depth interval by reach for this calculation. In deeper intervals, when the native material was reached in a given core, the polygon area associated with that core was removed from the area total for that reach and depth (as well as for all deeper depths). Therefore, this polygon-based approach accounted for the spatial variations in sediment thickness, as shown in Figure 3-2. The surface area of sediment by reach and depth interval is listed in Table 6-9.

The resulting surface water, surface sediment, and subsurface sediment preliminary chemical mass inventory estimates for TPAH (17), TPCB, and Cu are presented in Table 6-6.

6.5.1.2 Chemical Source Terms

Annual chemical mass loads for sources entering the Study Area were developed as follows:

- **Point sources**. The annual point source loads used in these comparisons were those presented in Section 5.1.4.3 (with additional detail in Section 3 of Appendix E); values were broken down by reach and by the following general point source categories: individually permitted stormwater and wastewater discharges, CSOs, WWTP treated effluent overflow, other stormwater discharges, and treated groundwater effluent. As discussed in Section 5.1.4.2, four methods were used to calculate point source chemical loads (representing a range of data treatment approaches). As such, point source load values for these mass balance evaluations are presented as ranges based on the minima (RUM/SOM hybrid) and maxima (CDM) values calculated from the four methods.
- **Groundwater**. The annual chemical loads by reach from groundwater that discharges vertically into the subsurface sediment used in these comparisons were those presented in Section 5.2.2 (with additional detail provided in Section 6 of Appendix F). As discussed in Section 4.9.3, due to uncertainty in partitioning coefficients, two methods were used to estimate dissolved phase concentrations for TPAH (17) and TPCB in groundwater (one based on Study Area-specific K_d values and one based on

literature K_{OC} values). As a result, the groundwater chemical loads to the subsurface sediment for TPAH (17) and TPCB are presented as a range in this section, consistent with Section 5.2.2 (see also Section 6.1 of Appendix F). Lateral groundwater discharge and shoreline seeps are additional potential sources of contaminants to the surface water (as discussed in Section 6.1); these were evaluated qualitatively using surface water data in this RI Report (as discussed in Section 6.4 of Appendix F) and will be the subject of further evaluation in the FS.[275]

- **Atmospheric deposition**. The annual chemical loads by reach due to atmospheric deposition used in these comparisons were those presented in Section 5.5.

The estimated annual chemical loads for these sources for TPAH (17), TPCB, and Cu are presented in Table 6-6.

6.5.1.3 Sediment/Water Interface Chemical Mass Exchange

As discussed in Section 6.4.3, sediment/water mass transfer can be separated into dissolved and particulate phase processes. Mass fluxes associated with these terms were estimated as follows:

- **Dissolved phase** chemical loads were calculated separately for surface porewater advection and diffusive exchange as follows:
    - The annual porewater advection load for a given reach was calculated using the spatial average groundwater seepage rate by reach (see Figures 5-24a through 5-24c),[276] surface area of the reach, and the arithmetic average 0- to 15-cm (0- to 6-inch) porewater concentration (see Section 4.8.2.2).
    - The annual porewater diffusive exchange load for each reach was calculated similarly, based on the difference in concentration between the arithmetic average dry weather dissolved phase surface water concentration (calculated from partition coefficients presented in Table 6-2b; see also Attachment E-C of Appendix E) and arithmetic average 0- to 15-cm (0- to 6-inch) porewater

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[275] During the development and calibration of the CFT model, chemical loadings from lateral groundwater discharge will be further evaluated through sensitivity analyses, the results of which will be presented and discussed in the CFT modeling report and summarized in the FS Report. Moreover, as discussed in Section 5.2, USEPA is planning a study to collect empirical data on lateral groundwater discharge that will support the FS.

[276] Consistent with the calculation of groundwater loads (see Section 5.2.2), net downward seepage fluxes were considered as zero for the purposes of this calculation.

concentrations, using an assumed porewater exchange coefficient of 10 cm/day (which is a typical value based on literature; e.g., Thibodeaux et al. 2001). As discussed previously and described in the MAM2 (Anchor QEA 2016c),[277] the porewater exchange coefficient is used to represent the net effect of multiple processes acting in unison, all of which result in a dissolved phase mass transfer at the sediment/water interface. Data from several contaminated sediment sites indicate that observed porewater mass transfer coefficients are typically much greater than those which would be expected from diffusion alone (e.g., Thibodeaux and Bierman 2003). As such, the porewater mass transfer coefficient is typically taken to be a site-specific parameter (e.g., USEPA 2000). Indeed, the value used for these calculations (10 cm/day) encompasses the effects of tidal exchange based on the typical gross tidal fluctuations observed during the USGS seepage studies (see Table 6-4 and Section 6.4.3.1.2).

- **Particulate phase** chemical exchange loads were estimated for net deposition. The annual net particulate phase chemical deposition loads were calculated by reach as the product of surface water particulate phase chemical concentrations, NSRs (converted from cm/year to a mass basis using the surface sediment dry densities in Table 6-8), and the surface area of the reach. The concentrations used in the calculation were based on the dry and wet weather surface water estimated/calculated particulate phase concentration arithmetic averages (see Tables 4-62, 4-64, and 4-68 for dry weather and Tables 4-65, 4-67, and 4-70 for wet weather). Due to uncertainty in the amounts of chemical mass deposition occurring during dry and wet weather (to be refined through the CFT modeling), a range of values was developed based on the wet and dry weather arithmetic average concentrations.[278] As discussed in Section 6.3.4, multiple LOEs were used to evaluate NSRs by reach (see Figure 6-5). The data sources used to select representative NSRs for each reach for the purposes of this evaluation are as follows:

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[277] As noted previously, the MAM2 was completed prior to the RI data evaluations. As such, refinement of the representation of porewater exchange, including the value for the porewater mass transfer coefficient, will occur as part of the CFT modeling effort.

[278] These ranges cannot be used to parse out how much of the load occurs under wet versus dry weather—doing so would not be possible based on empirical data alone, but the CFT model being developed for the FS could be used for this purpose.

- Main stem reaches: 1991 to 2012 differential bathymetry analysis
- English Kills, East Branch, and Maspeth Creek: 1999 to 2012 differential bathymetry analysis
- Dutch Kills: Pb-210 and Cs-137 geochronology analysis
- Whale Creek: Estimated based on NSRs observed in adjacent portions of the main stem

The resulting net sediment deposition loads (in MT/year) used in the calculation based on these representative NSRs are presented in Table 6-10.

The estimated annual chemical loads for these dissolved and particulate phase sediment/water exchanges for TPAH (17), TPCB, and Cu are presented in Table 6-6.

6.5.1.4 Subsurface/Surface Sediment Mass Transfer Processes

As discussed in Sections 6.4.4 and 6.4.5, chemical mass exchange between the surface sediment (i.e., top 15 cm [6 inch]) and subsurface sediment can occur as a result of two primary processes: burial and porewater advection.[279] Mass fluxes associated with these terms were estimated as follows:

- **Burial**. The annual chemical mass load associated with burial of surface sediment was calculated using the same method as described previously for deposition, except that the concentration used in the calculation was that of the surface sediment arithmetically averaged by reach (i.e., Tables 4-13, 4-16, and 4-18). Based on the principle of mass conservation, the mass of sediment added to the top of the surface sediment layer as new deposition results in the same mass of sediment leaving from the bottom by burial (which adds mass to the subsurface sediment). However, the concentrations associated with the two mass fluxes differ.
- **Subsurface porewater advection**. The annual porewater advection loads entering the surface sediment from the subsurface sediment were calculated using the same

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[279] It is recognized that diffusive and dispersive transport will occur within the subsurface sediment porewater and at the interface between subsurface and surface sediment; however, those fluxes have not been quantified in this analysis, because diffusive and dispersive mass transfer rates are much lower than advective rates. Diffusive and dispersive transport throughout the sediment bed will be quantified as part of the CFT model.

approach as described previously for surface porewater advection, except that porewater concentrations representative of the 15- to 30-cm (6- to 12-inch) depth interval (see Section 4.8.2.3) were used.[280]

The estimated annual chemical loads for these two processes for TPAH (17), TPCB, and Cu are listed in Table 6-6.

6.5.2 Comparison of Mass Load and Inventory Estimates

The chemical mass load and preliminary mass inventory estimates for the various terms described in the preceding subsection are compared by reach in Table 6-6. Study Area totals of the results from this evaluation are compared on diagrams for TPAH (17), TPCB, and Cu in Figures 6-40, 6-41, and 6-42, respectively. The relative importance of the sources and chemical transport pathways evaluated in this RI can be evaluated by comparing the relative magnitudes of the preliminary mass inventory and annual load estimates presented in these figures. Key observations for TPAH (17), TPCB, and Cu are provided in Sections 6.5.2.1 through 6.5.2.3.

Additional discussions of these mass balances are provided in the context of the CSM in Section 8. Furthermore, as discussed previously and described in the MAM2 (Anchor QEA 2016c), a CFT model that integrates the Study Area data in a quantitative mass balance framework is being developed for the FS. This model will combine the various processes described in Section 6.5.1, incorporating the effects of hydrodynamics, sediment transport, flux from sediment to surface water, and loads from point sources and groundwater. The model will be calibrated to Study Area data and will provide a refined quantification of many of the mass load estimates presented in this section.

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[280] Since the number of 0- to 15-cm (0- to 6-inch) porewater samples (53 locations) is much greater than the 15- to 30-cm (6- to 12-inch) samples (17 locations), and because the data from the two depths are generally well correlated and fairly similar (see Figures 4-206, 4-208, and 4-209), a scale factor based on the average ratio of the concentrations was used to calculate the average 15- to 30-cm (6- to 12-inch) porewater concentrations by reach from the average 0- to 15-cm (0- to 6-inch) porewater concentrations. The calculated scale factors were 1.46, 1.44, and 0.95 for TPAH (17), TPCB, and Cu, respectively.

6.5.2.1 TPAH (17)

The surface water of the Study Area represents a transient condition, given that the gross fluxes that add or remove mass are two to three orders of magnitude greater than the arithmetic average mass of TPAH (17) present in the surface water at any point in time (approximately 0.7 kg).

As shown in Figure 6-40, the largest loads affecting the surface sediment are dissolved phase transport of porewater to surface water, burial to the subsurface, and deposition from the surface water (which is derived from a spatially varying combination of loads from point sources and the East River[281]). These terms are generally of similar magnitudes and sum to a net annual reduction of TPAH (17) mass in the surface sediment, which is consistent with the decreases in concentration over time evidenced by the sediment core data (see Sections 4.3.3 and 6.4.4.5). The net mass reduction from these terms of about 280 kg/year (gross values are about 330 kg/year reduction and 40 to 70 kg/year gain), which equates to about 8% per year of the preliminary TPAH (17) mass inventory estimate in the surface sediment.

The subsurface sediment represents a large inventory of TPAH (17), which is relatively static compared to the surface sediment. The preliminary mass inventory estimate in the subsurface sediment is approximately 400 times greater than that of the surface sediment, which is attributed to the greater thickness, as well as the higher TPAH (17) concentrations associated with the long and dynamic history of chemical loads and deposition represented by the subsurface sediment (see Section 6.4.5.1). The annual load of TPAH (17) from groundwater entering the subsurface sediment (830 to 1,500 kg/year[282]) represents a relatively small portion of that inventory (0.1% or less). Moreover, the load associated with porewater advection exiting the subsurface sediment (20 kg/year; based on measured porewater concentrations at the 15- to 30-cm [6- to 12-inch] depth interval) is much smaller than the incoming

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[281] The exact proportions of chemical deposition load from point sources and the East River cannot be determined based on data alone, due to lack of empirical East River net load estimates; this will be quantified through the CFT modeling being developed to support the FS.

[282] As discussed in Section 5.2.2.1 (see also Section 6.1 of Appendix F), the range of TPAH (17) loads of 830 to 1,500 kg/year may be an overestimate, due to the possible presence of NAPL in a few groundwater samples with relatively high calculated dissolved TPAH (17) values.

groundwater load, which demonstrates the substantial sorption and attenuation provided by the subsurface sediment (as discussed in Section 6.4.5.3).

Finally, it is important to recognize that these mass loads and preliminary inventory estimates vary considerably by reach (see Table 6-6). Notable reach-by-reach variations include the following:

- The CM 2+ reach accounts for approximately 90% of the TPAH (17) groundwater load[282] and 60% of the subsurface sediment TPAH (17) inventory. However, as discussed in Section 3.2, due to the long and dynamic history of chemical loads and deposition processes within the Study Area, the spatial distribution of that mass inventory cannot be definitively linked to proximate upland sites or sources (including point sources).
- Almost one-third of the total point source load for TPAH (17) to the Study Area is accounted for by one treated groundwater effluent discharge in CM 0 – 1.
- The total loads for porewater transport from surface sediment to surface water are mostly accounted for by English Kills (approximately 50%) and CM 2+ (approximately 40%).
- The largest chemical deposition load occurs in CM 0 – 1, due to its much higher NSR.

6.5.2.2 TPCB

Based on the TPCB mass load and inventory comparison in Figure 6-41, many of the same observations noted in Section 6.5.2.1 for TPAH (17) can be made. Specifically, the surface water mass is transient (i.e., inventory much lower than gross fluxes), and the surface sediment is more dynamic than the larger, stable inventory in the subsurface sediment. Likewise, there is net annual reduction in the surface sediment inventory, reflecting the dilution effects of the ongoing natural recovery driven by deposition and mixing processes throughout much of the Study Area, as shown by the sediment core data.

However, there are differences between TPCB and TPAH (17) as well. First, the annual mass loads entering and leaving the surface sediment represent a smaller fraction of the preliminary surface inventory estimate (less than 5%) for TPCB as compared to TPAH (17) (i.e., 8%). The rate of turnover for TPCB is somewhat slower than that for TPAH (17), indicating that the relative importance of ongoing sources in controlling surface sediment concentrations is greater for TPAH (17) than TPCB. Second, the range of groundwater loads entering the subsurface sediment is among the smallest chemical mass transport terms quantified for TPCB, indicating groundwater is an unimportant transport mechanism for this chemical group.

Notable reach-by-reach variations in the TPCB mass load comparison (see Table 6-6) reflect the concentration distributions discussed in Sections 4 and 5, and include the following:

- The majority of the mass in both the surface and subsurface sediment is present in English Kills and CM 2+ (approximately 80% combined).
- English Kills and CM 2+ also account for most (approximately 90% combined) of the surface porewater TPCB flux (advection and diffusion combined).
- The largest point source loads are generally in the tributaries with large point source flows (e.g., English Kills, East Branch, Maspeth Creek, and Dutch Kills account for between one-half and two-thirds of the total point source TPCB load).
- Elevated per acre masses of PCBs in surface sediment occur at discrete locations within the Newtown Creek system, particularly within Dutch Kills, English Kills, and CM 2+ (see Table 6-6); associated upland sources have not been identified.

6.5.2.3 Cu

The mass load and inventory comparison for Cu (see Figure 6-42) shows many of the same observations as described previously for TPAH (17) and TPCB. Cu mass estimates indicate a transient surface water inventory, a relatively large and stable inventory in subsurface sediment, and smaller, somewhat more dynamic inventory in the surface sediment (with a gross turnover of about 3% per year, indicating a condition of decreasing mass and concentration over time). Particulate phase processes of deposition and burial represent the largest transport terms; groundwater loads and dissolved phase transport processes are relatively minor.

The Cu load and inventory estimates vary by reach as well. Notable spatial variations include the following:

- Many of the terms (e.g., mass in surface and subsurface sediment, groundwater load, and porewater advection and diffusive exchange loads) are greatest in CM 2+ and English Kills.

- The Cu point source loads are highest in the tributaries with the largest point source flows—East Branch, Maspeth Creek, and English Kills.
- Most of the Cu deposition load (approximately 60% of the Study Area total) occurs in CM 0 – 1, due to the high NSR in that reach.

6.6 Bioaccumulation

Bioaccumulation is the process by which chemicals accumulate in biological tissues, and biomagnification is the process by which chemical concentrations may increase with each trophic level, potentially reaching higher concentrations on a volumetric or mass basis compared with sediment and water. This section focuses on the bioaccumulation of TPCB, because the BHHRA and BERA identified PCBs as the primary chemical of concern via food ingestion, and because PCBs are bioaccumulative.[283]

Bioaccumulation represents the final step of the process by which contaminants move from their sources to endpoints that are of primary importance for decision-making. In this case, the endpoints are the upper trophic levels of the food web and human consumption of fish and crab. Fate and transport (see Section 6.4) focuses on the pathways of the contaminants from original sources (e.g., point sources, East River, industrial activities, chemical spills, vessel discharges, and other non-point sources) to surface sediment and surface water; bioaccumulation focuses on the pathways from surface sediment and surface water to the food web.

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[283] As noted in Section 4.1.2.2, in addition to PCBs, total dioxins/furans are a contributor to both cancer risk and noncancer hazards in crab tissue. Furthermore, PCBs and dioxins/furans are bioaccumulative. The discussion in this section focuses on PCBs, because dioxins and furans have distributions in surface sediments and tissue that are generally similar to those of PCBs (see Section 4.12), with few exceptions, and because their partitioning behavior is similar to PCBs. Furthermore, dioxins and furans and PCBs exhibit similar site signatures: for blue crab muscle+hepatopancreas TEQ 2005 (mammal), the ratio of the median in the Study Area to the median in the four reference areas is 4.1 for total dioxins/furans and 6.0 for PCBs (see Table 6-38 of the BHHRA). These values are within 50% of each other, and moreover, the higher value for PCBs suggests the possibility that the Study Area contribution to PCB body burdens may be somewhat greater than for dioxins/furans. Finally, key determinants of the Study Area contribution to biota body burdens (and, therefore, body burden response to remediation) are diet and movement patterns, which are the same for all chemicals. Therefore, evaluation of PCB patterns in tissue and bioaccumulation are sufficient for the purposes of this RI, noting that tables and figures are presented for all analytes in Appendix A. Nevertheless, for the evaluation of remedial alternatives in the FS, empirical data on surface sediment and/or tissue will be used to develop an approach for evaluating the extent to which each alternative may reduce risk from dioxins/furans, in addition to TPCB.

Spatial patterns in TPCB concentrations in the species that were sampled during Phase 2 are discussed in Section 4.10. Striped bass, blue crab, mummichog, Atlantic menhaden, white perch, and benthic invertebrates were sampled, because they represent a range of feeding guilds and were found in sufficient numbers during the Phase 1 and Phase 2 community surveys. Striped bass and blue crab are the primary species consumed by recreational anglers and crabbers, while Atlantic menhaden, mummichog, and benthic invertebrates represent components of their food web. White perch was added as a substitute for spot (*Leiostomus xanthurus*), which was initially included as a species for evaluation based on the Phase 1 community survey but was not found in sufficient quantities during Phase 2 sampling. In this subsection, the discussion is extended to focus on the relationships between PCB concentration in tissue of these species and exposure sources, to better understand this linkage.

6.6.1 Lipid Normalization

Organic compounds accumulate in fatty tissue of biota, and sometimes tissue concentrations are positively correlated with lipid content. When evaluating spatial patterns or time trends in such cases, it is important to control for the variability in lipid content among individuals. This is usually done by normalizing chemical concentrations by the amount of lipid (that is, dividing wet weight-based tissue concentrations by lipid content: [mg/kg wet weight]/[kg lipid/kg wet weight] = [mg/kg lipid]).

The potential value of lipid normalization for the Study Area was explored in two ways: 1) by evaluating the relationship between lipid and PCBs; and 2) by evaluating the extent to which lipid normalization reduces the variance in tissue PCB concentrations. The relationships between TPCB concentrations and lipid content in Study Area biota (striped bass, blue crab, mummichog, Atlantic menhaden, and white perch were sampled for the RI; see Section 7) are presented in Figures 6-43 through 6-47. The coefficient of variation (CV) (i.e., Sd divided by the arithmetic average) is also presented for both wet weight-based and lipid-normalized tissue TPCB concentrations for each species, for the Study Area and four Phase 2 reference areas (see Table 6-11). Lipid normalization would be considered beneficial if it reduced the CV.

There are no strong, consistent relationships between lipid content and TPCB for any species (e.g., r^2 values range from less than 0.01 to 0.07 in striped bass fillet and striped bass whole-body samples, respectively). The relationship is somewhat stronger for white perch ($r^2 = 0.52$; see Figures 6-43 through 6-47). An r^2 of 0.52 means that slightly more than half of the variance in PCBs can be explained by the lipid content of white perch. In addition, CVs for wet weight concentrations are similar to, or lower than, those for lipid-normalized concentrations (see Table 6-11). Despite the lack of a clear benefit to lipid normalization, spatial patterns and comparisons with reference areas for tissue samples are presented in the remainder of Section 6.6 on a wet weight and lipid-normalized basis (at the request of USEPA), for completeness and to maintain consistency with the MAM3 (Anchor QEA 2018).

6.6.2 PCB Concentrations in Tissues and in Exposure Sources

6.6.2.1 Resident Organisms

Bioaccumulation in resident organisms with limited home ranges can often be quantified using simple ratio or regression approaches. In Newtown Creek, the resident organisms collected and evaluated are benthic invertebrates and mummichog. Benthic invertebrates are exposed to sediment in the localized area where they are collected. Mummichog generally occupy a small home range of 36 to 38 meters along the banks of tidal creeks but have been documented moving as much as 380 meters (Lotrich 1975; USFWS 1985; Skinner et al. 2005; Teo and Able 2003). Over their home range, they are potentially exposed to chemicals in both surface sediment and surface water, in their food, and directly via gill exchange with surface waters.

For both of these organisms, several relationships were explored, including linear regression, log-log regression, and biota-sediment accumulation factor (BSAF) relationships. Based on evaluation of the slopes, correlation coefficients, and significance levels of these different relationships (presented in Sections 3 and 4 of the MAM3; Anchor QEA 2018), a simple BSAF relationship between tissue and sediment PCB concentration was used[284]:

$$BSAF = \frac{C_b}{C_s} \tag{6-1}$$

where:

$BSAF$ = biota-sediment accumulation factor (kg sediment/kg tissue, where sediment may be in dry weight or OC-normalized units and tissue may be in wet weight, dry weight, or lipid-normalized units)

C_b = chemical concentration in the tissue of biota (mg/kg wet weight, dry weight, or mg/kg lipid)

C_s = chemical concentrations in the sediment (mg/kg dry weight or mg/kg OC)

The BSAF relationship for benthic invertebrates was evaluated using the results of the 28-day laboratory bioaccumulation studies conducted using sediment collected from the Study Area and polychaetes of the species *Nereis virens*. Results for TPCB are presented in Figure 6-48, which shows the relationship using sediment on a dry weight basis and tissue on a wet weight basis.[285] In this figure, colors represent data collected in various reaches of the Study Area. There is a linear relationship between TPCB concentrations in sediment and benthic invertebrate tissue shown in Figure 6-48. The regression of the linear relationship with the intercept forced through the origin matches the data well, indicating that a simple BSAF relationship is appropriate. Figure 6-48 shows that for benthic invertebrates, tissue

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[284] The term *BSAF* refers to the use of a single value, or multiplier, to describe the relationship between sediment and tissue contaminant concentration. Mathematically, the use of a BSAF assumes a linear relationship with an intercept of zero (i.e., tissue = constant × sediment). That is, tissue concentrations follow sediment concentrations linearly. For example, for every doubling of sediment concentrations, tissue concentrations double. An intercept greater than zero indicates that tissue concentrations do not fully follow sediment concentrations. In the extreme case, a slope of zero would mean that tissue concentrations bear no relationship to local sediment. An intercept greater than zero (assuming it is not just due to data uncertainty) could be caused by exposure to all the sediment sources used in the analysis (so that the organism in effect integrates over all sediment) or the presence of other chemical sources (e.g., surface water exposure or exposure to a wider home range than represented by the sediment data).

[285] An expanded version of this figure was developed in the MAM3 (Anchor QEA 2018), in which all possible combinations of units that are available from the project data were compared (i.e., sediment on a dry weight or OC basis and tissue on a wet weight, dry weight, or lipid basis). Results from those other combinations of units were similar to those in Figure 6-48.

concentrations are strongly tied to the sediment concentrations, which accounts for 98% of the variability in the data. Thus, this analysis demonstrates that the relationship between local sediment on a dry weight basis and invertebrate tissue on a wet weight basis is sufficient to describe this exposure linkage for evaluating bioaccumulation.

The relationship for mummichog was characterized using field-collected data. Each mummichog sample was matched with the surface sediment data within a specified distance of its collection location. Home range diameters from 19 to 380 meters were evaluated, consistent with published information concerning the home range of this species (Lotrich 1975; USFWS 1985; Skinner et al. 2005). At the request of USEPA, results for TPCB in mummichog versus sediment for each home range are presented in Figure 6-49; the top row shows tissue on a wet weight basis and sediment on a dry weight basis, and the bottom row shows tissue on a lipid basis and sediment on an OC basis. An evaluation of the various home range scenarios (presented in Section 3.3 of the MAM3; Anchor QEA 2018) indicated that a home range of 160 meters best characterized the mummichog/sediment relationship. There is a positive relationship between TPCB concentration in mummichog tissue and local surface sediment (see panels labeled 160 meters in Figure 6-49). This is particularly evident in Dutch Kills, where two of the mummichog samples were collected from a location with relatively lower sediment TPCB concentrations and two were collected from a location with higher concentrations. The tissue concentrations increase with the sediment on a local basis, even within this one reach of the Study Area. However, the slope (0.60; $p < 0.001$) of the log-log relationship is somewhat lower than for benthic invertebrates, suggesting that there may be other exposure sources to mummichog, in addition to local surface sediment (e.g., surface water or different home range sizes). Potential causes for the deviation from a slope of 1.0 will be evaluated in the FS based on empirical data.

6.6.2.2 Non-Resident Organisms

Bioaccumulation in non-resident, mobile organisms that migrate may not be quantified by a simple BSAF approach. In Newtown Creek, striped bass, blue crab, Atlantic menhaden, and white perch exhibit wide-ranging movement. Variation in migration and movement patterns is an important consideration for remedial decision-making, including exposure outside Newtown Creek, within the New York Harbor area. In addition, exposure through dietary items may not be captured by local sediment concentrations alone. For example, Atlantic menhaden primarily feed in the water column, whereas striped bass and blue crab forage on a combination of dietary items tied to both the sediment and water column. Their natural history is described in Attachment F of the BERA (see Appendix I) and in Appendix A of the MAM3 (Anchor QEA 2018). For ease of comparison to resident organisms and presentation of tissue concentrations relative to local sediment concentrations, the BSAF model is shown for these non-resident species, although this is not the presumptive model for evaluating bioaccumulation of PCBs in these non-resident species.

Wide-ranging migratory behavior is reflected in TPCB tissue concentrations. This means that a BSAF model with Study Area sediment is unlikely to realistically represent all of the TPCB exposure of striped bass. Figure 6-50a presents the relationships between arithmetic average dry weight surface sediment TPCB concentration and wet weight and lipid-normalized whole-body tissue concentrations for striped bass measured in each of the four Phase 2 reference areas and in each FSZ in the Study Area. The concentrations in the reference areas are lower than in the Study Area, both for sediment and striped bass whole-body samples (on both wet weight and lipid-normalized basis), indicating a site-specific contribution to body burdens. However, the relationship between surface sediment and tissue concentrations is weak, as evidenced by the negative r^2 values,[286] and appears to be driven by the fact that the reference area concentrations as a whole are lower than the Study Area concentrations as a whole. That is, there is no clear relationship between surface sediment and tissue concentrations within the Study Area (i.e., as sediment concentrations increase within the Study Area, the tissue concentrations do not change). Likewise, there is no clear relationship between surface sediment and tissue concentrations within the reference areas. Similar conclusions can be drawn from the striped bass fillet data (see Figure 6-50b). Although the fact that there is an overall relationship indicates that some of the body burden comes from Study Area sediments, the lack of a linear relationship with the intercept through zero for sediment and striped bass indicates that Study Area sediments

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[286] A negative r^2 value for this form of regression can result when the intercept of the regression line is forced through zero (i.e., the equation does not contain a constant term), but the data indicate that the intercept is actually significantly different from zero. In this case, forcing the regression line through zero also forces the line to be far from the data. Therefore, a negative r^2 indicates that the regression line is not a good fit for the data, and a BSAF relationship is not appropriate for these data.

alone do not fully explain the PCBs in striped bass; rather, the PCBs in striped bass represent a mix of exposure sources, consistent with this species' life history and diet. This lack of a relationship is diagnostic of a wide-ranging species; the fish collected in any one location reflect their exposure history over a wider area, rather than just the local sediment.

Similar patterns are observed for blue crab (see Figure 6-51a [whole body] and Figure 6-51b [muscle and hepatopancreas]); within the Study Area, there is no clear relationship between surface sediment and tissue. In contrast to striped bass, both sediment and tissue concentrations are lower for blue crab collected in the three Jamaica Bay reference areas than for those collected in Westchester Creek. This spatial pattern in PCB concentrations suggests that, although blue crab are wide-ranging, they probably spend more time within each waterbody than striped bass (i.e., there is probably somewhat more site fidelity).

Similar to striped bass and blue crab, Atlantic menhaden and white perch exhibit patterns indicating widespread movement and exposure over areas much larger than the Study Area (see Figures 6-52 and 6-53).

In contrast to the case with resident organisms, the fact that individual non-resident organisms collected in the Study Area are part of larger regional populations, with ranges much larger than Newtown Creek and its tributaries, means that reductions in sediment PCB concentrations within the Study Area through natural recovery and remedial actions are unlikely to result in parallel reductions in chemical concentrations in the tissues of these organisms.

7 RISK ASSESSMENT SUMMARY

This section summarizes the BHHRA and BERA, conducted to assess risks posed to humans and ecological receptors due to exposure to CERCLA hazardous substances in media (e.g., water and sediment) within the Study Area under current and future conditions assuming no remediation. The BHHRA and BERA were conducted to support the RI, following USEPA guidance. The complete BHHRA and BERA are presented in Appendices H and I, respectively.

Risk assessments are conducted to determine whether there are risks to human health and the environment from exposure to site-related releases of hazardous substances. If risks are present, these assessments are used to inform risk-based management decisions regarding remedial actions that may be required to mitigate unacceptable risks. Risk assessments serve as a basis with which to compare alternatives developed in the FS for remediation of contaminated media in accordance with the requirements in Section 300.430(e)(9)(iii) of the NCP (USEPA 1990b).

7.1 Human Health Risk

Section 7.1 summarizes the BHHRA (see Appendix H), which evaluates risks to human health associated with both current and potential future human exposures to CERCLA hazardous substances present in the Study Area in the absence of control or mitigation actions. The BHHRA presents risk estimates for a number of exposure scenarios in which people could potentially be exposed to chemicals found in surface sediment, surface water, fish and crab tissue, and air in the Study Area. The data used in the BHHRA consist of surface sediment, surface water, tissue, and air data collected during Phase 1 and Phase 2 of the RI.

The BHHRA concludes that the only recreational receptor categories and exposure pathways with estimated cancer risks above the USEPA acceptable cancer risk range of 10^{-6} to 10^{-4} (i.e., up to 8×10^{-4}) and noncancer hazard index (HI) greater than the threshold of 1 (i.e., up to 40) result from the consumption of fish and crab tissue by recreational anglers and crabbers. For all other recreational receptor categories and exposure pathways, cancer risks are within or below USEPA's acceptable risk range, and noncancer hazards are below the hazard threshold. The general construction worker is the only occupational receptor category with noncancer hazards above the hazard threshold of 1 (i.e., 2). Cancer risks for the general construction worker are within USEPA's acceptable risk range of 10^{-6} to 10^{-4} (i.e., 2 x 10^{-6}); for all other occupational receptor categories and exposure pathways, cancer risks are also within or below USEPA's acceptable risk range.

For the recreational consumption of fish and crab exposure scenario, total nondioxin-like PCB congeners, TPCB congener TEQ, and total dioxin/furan TEQ are the COPCs associated with these risks and hazards. For the general construction worker, the HIs were less than 1 for all target organs except for the central nervous system (i.e., 1.1). The HI for potential central nervous system effects was primarily due to PCBs in surface sediment.

The basic steps of the CERCLA human health risk assessment process are as follows:

- Data collection and evaluation
- Exposure assessment
- Toxicity assessment
- Risk characterization
- Uncertainty analysis

A summary of each basic step is provided in this section.

7.1.1 Data Collection and Evaluation

This section briefly summarizes the data evaluation and screening process, which identifies COPCs to be evaluated in the baseline risk assessment. Additional details regarding these topics are presented in Section 3 of the BHHRA (see Appendix H). The dataset used in the BHHRA consists of surface sediment, surface water, fish and crab tissue, and air sample results. Environmental investigations to support the BHHRA were conducted from 2010 to 2014. The BHHRA dataset includes sample data and matrices needed to quantitatively evaluate the potential human health exposures. Data from the Phase 2 reference areas are also used to evaluate the relative magnitude of risks in the Study Area. The number of samples from each medium from the Study Area and Phase 2 reference areas evaluated in the BHHRA are as follows:

- 399 surface sediment samples from the Study Area

- 40 surface sediment samples from the Phase 2 reference areas
- 362 surface water samples from the Study Area
- 31 surface water samples from the Phase 2 reference areas
- 27 tissue composite samples from the Study Area
- 45 tissue composite samples from the Phase 2 reference areas
- 24 ambient air samples from the Study Area
- 5 ambient air samples from upland background locations near the Study Area

The COPC screening process was conducted for surface sediment, surface water, biota tissue (i.e., fish and crab), and air. The COPC selection process for all media evaluated is consistent with USEPA guidance and is used to identify chemicals detected in the Study Area that are most likely to contribute significantly to human health risk (USEPA 1989, 1993). Consistent with the COPC screening process, the SLs are based on conservative, default exposure assumptions. The SLs for each medium are described here:

- **Surface sediment**. The SLs are the USEPA Residential Soil Regional Screening Levels (RSLs; USEPA 2015c). Human health SLs are not available for sediment exposures, so soil SLs are used as a conservative surrogate.
- **Surface water**. The SLs are the USEPA tap-water RSLs (USEPA 2015c).
- **Biota tissue**. The SLs are USEPA's fish tissue RSLs (USEPA 2015c). The fish tissue RSLs are based on an adult recreational angler using a mean fish consumption rate of 54 grams per day, which was derived from a 1982 United States Department of Agriculture study (Pao et al. 1982; USEPA 1991).
- **Air**. The SLs are the residential air RSLs (USEPA 2015c).

Consistent with USEPA CERCLA Risk Assessment Guidance, the cancer risk level for the RSLs is based on a 1×10^{-6} risk level, and for noncancer hazards, an HQ of 0.1.

As a result of the screening process, the chemicals were placed into one of the following three categories:

- **COPC**. Chemicals with an FoD greater than 5% and a maximum detected concentration above their SLs were carried forward and evaluated further in the

BHHRA. In addition, all Group A carcinogens detected in site media were retained as COPCs, regardless of the FoD and availability of SLs for those chemicals.

- **Eliminated as COPC**. Chemicals with concentrations below their respective SLs or chemicals with an FoD less than 5% were eliminated as COPCs and were not evaluated in the BHHRA.
- **Uncertain COPC**. Chemicals without an SL, but with an FoD greater than 5%; or chemicals with an FoD less than 5%, but with an RL greater than their respective SLs, are evaluated in the uncertainty section of the BHHRA (see Section 7 of Appendix H).

The COPCs for each medium identified based on the SL analyses are presented in Table 7-1.

7.1.2 Exposure Assessment

This section summarizes the CSM, selection of exposure pathways, selection of exposure scenarios, and associated exposure parameters used in the BHHRA. Additional details regarding these topics are presented in Section 4 of the BHHRA (see Appendix H).

The predominant land uses around Newtown Creek and its tributaries are industrial, with pockets of mixed use, commercial, and high-rise residential developments near the East River. As a result of these land uses, access to the Study Area by the public is restricted on the land side by physical controls (e.g., fences) and security/surveillance controls maintained and operated by the industrial facilities along Newtown Creek. Notwithstanding these access restrictions, there are areas along Newtown Creek where people can gain access to Newtown Creek; existing public access areas in Newtown Creek and its tributaries are summarized in Section 3.2.7, including the types of recreational activities that may and do occur at these locations. The access restrictions present in the Study Area do significantly limit the public's ability to engage in recreational activities along much of the shoreline of Newtown Creek.

Based on the current and future uses of the Study Area, five categories of recreational users, four categories of occupational users, one category of unauthorized users (sailboat users), and one general exposure scenario involving residents and occupational workers (local flooding scenario) were identified for quantification of risks for the BHHRA. These categories and the respective exposure pathways for each category are as follows:

- Recreational users
  - Boaters/swimmers: assessed potential dermal contact with and incidental ingestion of surface water and inhalation of ambient air exposure pathways. The dermal contact and incidental ingestion of surface sediment were qualitatively evaluated in the BHHRA, as these exposure pathways are considered complete but with low exposure potential.
  - Recreational anglers and crabbers: assessed potential dermal contact with surface water, inhalation of ambient air, and the ingestion of fish and crab tissues exposure pathways. Incidental ingestion of surface water was qualitatively evaluated in the BHHRA, as this exposure pathway is considered complete but with low exposure potential.
  - Plank Road Area recreational users: assessed potential dermal contact with surface water, dermal contact and incidental ingestion of surface sediment, and inhalation of ambient air exposure pathways (USEPA 2014c). The incidental ingestion of surface water was qualitatively evaluated in the BHHRA, as this exposure pathway is considered complete but with low exposure potential. While this exposure scenario is specific to activities that may occur at the Plank Road Area, this type of recreational activity that includes limited exposure to nearshore sediment and surface water may be occurring at some of the access locations described in Section 3.2.7. The results of the evaluation of Plank Road Area recreational users can be applied to these other access areas.
  - Shoreline recreational users: assessed potential inhalation of ambient air exposure pathway.
- Occupational users
  - Landside workers: assessed potential inhalation of ambient air exposure pathway.
  - Dockside workers: assessed potential dermal contact with and incidental ingestion of surface water, dermal contact and incidental ingestion of surface sediment, and inhalation of ambient air exposure pathways.
  - Future Hunter’s Point South construction workers: assessed potential dermal contact with and incidental ingestion of surface water, dermal contact and

  incidental ingestion of surface sediment, and inhalation of ambient air exposure pathways (USEPA 2014d).
  - General construction workers: assessed potential dermal contact with and incidental ingestion of surface water, dermal contact and incidental ingestion of surface sediment, and inhalation of ambient air exposure pathways (USEPA 2014d).
  - The BHHRA estimates risks posed by a site in the absence of remediation or exposure controls. Thus, the BHHRA does not assume the use of worker protections and criteria regarding chemical contamination that might be required under health and safety regulations.

- Unauthorized users
  - Sailboat users: assessed potential dermal contact with surface water, dermal contact and incidental ingestion of soil/fill, and inhalation of ambient air exposure pathways (USEPA 2014d). Surface sediment samples collected adjacent to the areas where the sailboats are moored serve as a surrogate to represent the soil/fill material. The incidental ingestion of surface water was qualitatively evaluated in the BHHRA, as this exposure pathway is considered complete but with low exposure potential. As described in Section 4.4.8.10 of the BHHRA, these sailboat users are visitors to illegally moored sailboats along the bulkheads near Vernon Boulevard in the Study Area (see Appendix H).
  - Trespassers/homeless people: qualitative evaluation.

- Flooding scenario
  - Residents and occupational workers: assessed potential dermal contact with and incidental ingestion of surface water and dermal contact and incidental ingestion of surface sediment exposure pathways (USEPA 2014d). Inhalation of ambient air during flooding would also occur and was evaluated qualitatively, due to the uncertainty in estimating air concentrations related to flooding events.

The receptor categories and exposure pathways for these receptors are presented in the BHHRA exposure pathway CSM figures for current and future conditions (see Figures 7-1 and 7-2, respectively). The BHHRA exposure pathway CSM figures describe potential contaminant sources, transport mechanisms, potentially exposed populations, exposure pathways, and routes of exposure. The CSM is used to identify potentially complete and incomplete exposure pathways, and for potentially complete exposure pathways, whether the pathways are to be evaluated qualitatively or quantitatively in the BHHRA.

CERCLA-based human health exposure assessments are conducted for both the reasonable maximum exposure (RME) and central tendency exposure (CTE), considering both current and potential future land use conditions (USEPA 1989). The RME is defined as the highest exposure that is reasonably expected to occur at a site, but that is still within the range of possible exposures. RME estimates are intended to avoid underestimating risks, but exposure (and, thus, risk estimates) may be overestimated for many individuals (USEPA 1989). USEPA generally uses RME scenarios to evaluate remedial actions at a site (USEPA 1989). The CTE is defined as a more typical (or average) estimate of exposure. In the BHHRA, CTE risks are only calculated for the receptors and exposure pathways that exhibited total cancer risk above USEPA's acceptable cancer risk range or total noncancer hazards above the threshold of 1 for the RME scenario. USEPA's acceptable cancer risk range is 1×10^{-6} to 1×10^{-4} as established in the NCP and further discussed in Section 7.1.4 (USEPA 1990b). Site-specific exposure parameters were used to evaluate RME and CTE scenarios whenever possible. These exposure parameters are conservative and are applicable throughout the Study Area.

7.1.3 Toxicity Assessment

The toxicity assessment step is the process of identifying toxicity values used in this risk assessment for individual chemicals and chemical classes. The toxicity values used in this risk assessment are the values recommended by USEPA and have been peer reviewed. Two types of adverse health effects are evaluated—potential incremental risk of developing cancer due to exposure to chemicals and the hazards associated with noncancer health effects due to exposure to chemicals. Additional details regarding toxicity are presented in Section 5 of the BHHRA (see Appendix H).

In accordance with USEPA guidance (2003b), the chronic toxicity values used were obtained from the following sources, according to the hierarchy outlined here:

- The Integrated Risk Information System (IRIS) database available through the USEPA Environmental Criteria and Assessments Office in Cincinnati, Ohio (USEPA 2015d).

IRIS, prepared and maintained by USEPA, is an electronic database containing peer-reviewed health risk and USEPA regulatory information on specific chemicals.

- USEPA Provisional Peer Reviewed Toxicity Values (PPRTVs; USEPA 2015e), provided by the Office of Research and Development, National Center for Environmental Assessment, Superfund Health Risk Technical Support Center, which develops these values on a chemical-specific basis when requested under the USEPA Superfund program.
- Other sources of information, with a preference for these sources: 1) those that provide toxicity information based on similar methods and procedures as those used for IRIS and PPRTV values; and 2) those that contain values that are peer-reviewed, available to the public, and transparent with respect to the methods and processes used to develop the values. Examples of recommended sources include, but are not limited to, the California Environmental Protection Agency (OEHHA 2016); the Agency for Toxic Substances and Disease Registry (ATSDR) Minimal Risk Levels (MRLs; ATSDR 2015), which represent estimates of the daily human exposure to a hazardous substance that is likely to be without appreciable risk of adverse noncancer health effects during a specified duration of exposure; and USEPA *Health Effects Assessment Summary Tables* (HEAST) toxicity values (USEPA 1997b).

COPCs were quantitatively evaluated on the basis of their carcinogenic and/or noncancer potential. The toxicity values used for evaluating exposure to chemicals with carcinogenic and noncancer effects were the slope factor (SF) and reference dose (RfD), respectively, for surface sediment, surface water, and tissue; or the Inhalation Unit Risk (IUR) and reference concentration (RfC), respectively, for air.

USEPA's *Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens* (2005c) was applied to evaluate excess cancer risk related to childhood exposure to mutagenic carcinogens, such as carcinogenic PAHs. Specifically, the evaluation of all mutagenic COPCs applied the recommended age-specific adjustment factors to the cancer SF.

Oral cancer SFs and IURs are used to estimate the risk of cancer associated with exposure to a carcinogen. The oral SF represents an upper bound, generally approximating a 95% UCL, on the increased cancer risk from a lifetime exposure by ingestion. SFs are expressed in units of proportion (of a population) affected (milligrams per kilograms per day [mg/kg-day])$^{-1}$. An IUR is an upper-bound excess lifetime cancer risk estimated to result from continuous inhalation exposure at a concentration of 1 microgram per cubic meter in air. The SFs developed by USEPA represent plausible upper-bound estimates, which means that USEPA is reasonably confident that the actual cancer risk will not exceed the estimated risk calculated using the SF (USEPA 1996, 2005d).

Noncancer health effects are evaluated using RfDs developed by USEPA. The RfD and RfC provide quantitative information for use in risk assessments for health effects known or assumed to be produced through a nonlinear (possibly threshold) mode of action. The RfD, expressed in units of mg/kg-day, is defined as an estimate of a daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime for chronic RfDs or during a portion of a lifetime (i.e., less than 1 year) for sub-chronic RfDs. The inhalation RfC, expressed in units of milligrams per cubic meter, is analogous to the oral RfD, but it provides a continuous inhalation exposure estimate and considers toxic effects for both the respiratory system (portal of entry) and effects peripheral to the respiratory system (systemic effects). The use of RfDs and RfCs is based on the concept that there is a range of exposures that exists up to a finite value, or threshold, that can be tolerated without producing a toxic effect. The BHHRA uses available chronic RfDs and RfCs for the oral and inhalation exposure routes, respectively.

In situations where exposures to human receptors are a 1-year duration or less, USEPA Superfund guidance allows for the use of sub-chronic RfDs and RfCs to evaluate noncarcinogenic hazards (USEPA 2002b). For the BHHRA, the construction workers at Hunter’s Point South and the general construction worker receptor fall under this category. For these receptors, the BHHRA uses available sub-chronic RfDs and RfCs for the oral and inhalation exposure routes, respectively. The sub-chronic RfD and RfC were obtained from the following sources according to the hierarchy outlined here:

- PPRTVs provided by the Office of Research and Development, National Center for Environmental Assessment, Superfund Health Risk Technical Support Center (USEPA 2015e)
- The ATSDR MRLs (ATSDR 2015)
- PPRTV Assessment Appendix screening toxicity values (USEPA 2015e)

- USEPA HEAST toxicity values (USEPA 1997b)
- The IRIS database available through the USEPA Environmental Criteria and Assessments Office in Cincinnati, Ohio (USEPA 2015d)

7.1.4 Risk Characterization

The risk characterization step in a BHHRA provides estimates of the magnitude of the potential adverse health effects caused by exposure to COPCs. The risk characterization step combines the information developed in the exposure and toxicity assessment steps to calculate cancer risks and noncancer hazards. Additional details regarding the risk characterization are presented in Section 6 of the BHHRA (see Appendix H).

Potential carcinogenic and noncancer health effects are evaluated separately in this risk assessment. Risk estimates represent the theoretical probabilities of developing cancer over a lifetime due to exposure to site-related COPCs (e.g., as one in a million for 1 x 10^{-6} or one in ten thousand for 1 x 10^{-4}). Cancer risk estimates are compared to USEPA's acceptable risk range of 10^{-6} to 10^{-4} established in the NCP (USEPA 1990b). The potential for noncancer hazards is represented by the ratio of the estimated chemical intake to the critical chemical dose (e.g., an RfD) and is expressed as an HQ. HQs are not risk probabilities; the likelihood of an adverse effect may not increase linearly as the HQ increases. An HQ greater than 1 only indicates potential adverse health effects from the chemical exposure. HQs for individual COPCs and exposure routes are then summed to calculate an HI. However, summing HQs for COPCs that differ in target organ and/or mechanism of action could overestimate the potential for adverse effects. Therefore, consistent with USEPA guidance (USEPA 1989), if an HI for an exposure pathway is greater than unity, target organ-specific HIs are calculated to indicate the potential for noncancer hazards from simultaneous exposures to several COPCs. The conclusions of this analysis are included in the risk characterization summaries in this section.

The conclusion of the BHHRA risk characterization is that the only unacceptable human health cancer risks or noncancer hazards were associated with recreational fishing and crabbing consumption and general construction work. For the following receptors, the potential cancer risks and noncancer hazards are below or within USEPA's acceptable cancer risk range and below USEPA's noncancer hazard threshold:

- Recreational users
  - Current/future recreational boaters
  - Current/future swimmers/bathers
  - Current/future shoreline recreational users
  - Future Plank Road area recreational users
- Occupational users
  - Current/future landside workers
  - Current/future dockside workers
  - Future construction workers at Hunter's Point South
- Unauthorized users
  - Current/future sailboat users
- Flooding scenario
  - Current/future residents and occupational workers

The estimated RME total cancer risks and total noncancer HIs for the receptors evaluated are summarized in Table 7-2. The only recreational receptors in which the estimated RME cancer risks are above USEPA's acceptable risk range of 1 x 10^{-6} to 1 x 10^{-4} or the estimated noncancer HIs are greater than the threshold of 1 are the recreational anglers and crabbers from the consumption of fish and crab tissue (i.e., cancer risks up to 8 x 10^{-4} and noncancer HIs up to 40). For occupational receptors and exposure pathways evaluated in the BHHRA, only the general construction worker exposure to surface sediment results in an exceedance of the USEPA noncancer HI threshold of 1 (i.e., HI = 2), and the total cancer risk estimate for the general construction worker is within USEPA's acceptable risk range.

The estimated RME cancer risks and noncancer hazards for striped bass fillet consumption by recreational anglers are shown in Table 7-3, along with a summary of COPCs that have estimated cancer risks above the USEPA acceptable risk range and noncancer hazards above the USEPA HI threshold. The estimated RME total cancer risks for the separate adolescent and child age classes of recreational anglers are within USEPA's acceptable risk range, but the estimated RME total cancer risks for the adult and combined adult/child age classes are above the USEPA acceptable risk range (i.e., cancer risks up to 3 x 10^{-4}). The estimated RME

HIs are greater than the threshold of 1 for the adult, adolescent, and child age classes (i.e., noncancer HIs up to 20). For the recreational consumption of striped bass fillet exposure scenario, the primary contributors to both cancer risks and noncancer hazards are total nondioxin-like PCB congeners and TPCB congener TEQ.

The estimated RME cancer risks and noncancer hazards for white perch fillet consumption by recreational anglers are shown in Table 7-4, along with a summary of COPCs that have estimated cancer risks above the USEPA acceptable risk range and noncancer hazards above the USEPA HI threshold. The estimated total cumulative RME cancer risks for the adolescent and child age classes are equal to the upper end of USEPA's acceptable risk range, but the estimated RME total cancer risk for the adult and combined adult/child are above the USEPA acceptable risk range (i.e., cancer risks up to 3 x 10^{-4}). The estimated RME HIs are greater than 1 for the adult, adolescent, and child age classes (i.e., noncancer HIs up to 20). For the recreational consumption of white perch fillet exposure scenario, the primary contributors to both cancer risks and noncancer hazards are total nondioxin-like PCB congeners and TPCB congener TEQ.

The estimated RME cancer risks and noncancer hazards for blue crab muscle and hepatopancreas consumption by recreational crabbers are shown in Table 7-5, along with a summary of COPCs that have estimated cancer risks above the USEPA acceptable risk range and noncancer hazards above the USEPA HI threshold. The estimated RME total cumulative cancer risks for the adult, adolescent, child, and the combined adult/child age classes are above USEPA's acceptable risk range (i.e., cancer risks up to 8 x 10^{-4}). The estimated RME HIs are greater than the threshold of 1 for the adult, adolescent, and child age classes (i.e., noncancer HIs up to 40). For the recreational consumption of blue crab muscle and hepatopancreas exposure scenario, the primary contributors to both cancer risks and noncancer hazards are total nondioxin-like PCB congeners, TPCB congener TEQ, and total dioxin/furan TEQ.

The estimated RME cancer risks and noncancer hazards for the general construction worker are shown in Table 7-6, along with a summary of COPCs that have estimated cancer risks above the USEPA acceptable risk range and noncancer hazards above the USEPA HI threshold. The total cumulative RME cancer risks from exposure to surface sediment, surface water, and ambient air are within USEPA's acceptable risk range for adults. For the adult general construction worker, the RME noncancer HI is greater than 1 (i.e., 2). For the RME general construction worker, the primary contributor to noncancer hazard is total nondioxin-like PCB congeners.

For the CTE scenario, the only receptors for which CTE cancer risks and noncancer hazards are calculated are the recreational angler/crabber receptor and general construction worker, because only these receptors show potential RME cancer risks above USEPA's acceptable risk range and/or noncancer HI above the threshold of 1. The estimated CTE cancer risks and noncancer hazards are presented in Table 7-7. For the recreational angler/crabber, the estimated CTE cancer risks are within USEPA's acceptable risk range of 1×10^{-6} to 1×10^{-4}, but the estimated CTE noncancer HIs for all age classes and tissue types are above the threshold of 1 (i.e., HIs ranging from 2 to 5). For the general construction worker, the estimated CTE cancer risks and noncancer hazards are less than USEPA's acceptable risk range and below the HI threshold of 1.

7.1.5 Study Area Risks Versus Phase 2 Reference Areas Risks

In some cases, the same hazardous substances, pollutants, and contaminants associated with a release within the Study Area are also present in reference areas that are not related to Study Area releases. These reference areas reflect regional natural and/or anthropogenic conditions that should be included in the risk assessment, particularly when their concentrations exceed risk-based concentrations. In cases where reference area levels are high or present health risks, this information may be important to the public. Reference area information is important to risk managers, because the CERCLA program should not perform active remediation to concentrations below natural or anthropogenic background levels (USEPA 2002c).

COPC concentrations in Phase 2 reference area fish and crab tissues were elevated. The Phase 2 reference area and Study Area data were compared to USEPA's acceptable cancer risk range and noncancer hazard thresholds (see Table 7-8).

As shown in Table 7-8, the estimated cancer risks and noncancer hazards for the Phase 2 reference areas are lower than those estimated for the Study Area for the consumption of fish and crab tissue. However, there are estimated Phase 2 reference area cancer risks that either exceed USEPA's acceptable risk range or are equal to the upper end of the acceptable risk range. All the Phase 2 reference area noncancer hazards exceed the USEPA HI threshold of 1. The consumption of striped bass tissue from the Phase 2 reference areas results in an estimated cancer risk that is equal to the upper end of USEPA's acceptable risk range for the adult/child age class, and the noncancer hazards are above the USEPA hazard threshold for the adult, adolescent, and child age classes. The consumption of white perch tissue from the Phase 2 reference areas results in an estimated cancer risk that exceeds the USEPA's acceptable risk range for the adult/child age class, and the noncancer hazards are above the USEPA hazard threshold for the adult, adolescent, and child age classes. The consumption of blue crab tissue from the Phase 2 reference areas results in an estimated cancer risk that exceeds the USEPA's acceptable risk range for the adult/child age class, and the noncancer hazards are above the USEPA hazard threshold for the adult, adolescent, and child age classes.

The species consumed by people in the Study Area and used to represent human exposure in this risk assessment—namely, striped bass, white perch, and (to a lesser extent) blue crab—exhibit wide-ranging movement and are exposed to contamination present in the wider New York-New Jersey urban area. Furthermore, the food web of striped bass, white perch, and blue crab species may also be wide-ranging or largely water column-based, meaning that the base of the food web (smaller fish, phytoplankton, and zooplankton) likely accumulates contaminants from outside, as well as within, the Study Area.

PCB is the primary COPC that contributed to both cancer risk and noncancer hazard estimates in the Study Area, and in the Phase 2 reference areas. These Phase 2 reference area results, along with an understanding of species migration and movement, indicate that fish and crab exposure to COPCs occurs on a regional scale, and COPCs in the species consumed by people fishing and crabbing in the Study Area likely originate in a wider regional urban area beyond just the Study Area boundaries. As summarized previously, these regional-scale cancer risks are in the upper end of (or above) the USEPA acceptable risk range and noncancer hazards exceed the HI threshold of 1. The cancer risks and noncancer hazards calculated for the Phase 2 reference areas provide one estimate of regional risks that could be present in the absence of Study Area-related contamination.

7.1.6 Uncertainty Analysis

The presence of uncertainty is inherent in the CERCLA risk assessment process and is related to whether the risk estimates calculated in a BHHRA accurately represent the potential risks experienced by the human receptors exposed to contaminants in various Study Area environmental media. There are uncertainties associated with many aspects of the risk assessment process, including decisions made regarding sampling and analysis of environmental media, exposure assumptions used to quantify risk to human receptors, and toxicity values used in the risk characterization process. One of the most important areas of uncertainty in a BHHRA is the conservative nature of the risk characterization process to be protective of human health. Consistent with USEPA CERCLA risk assessment guidance, the RME exposure assumptions were developed to result in high-end estimates of the risks associated with the Study Area. To be health protective of potentially exposed populations, these risk estimates are intended to not underestimate risk, so they are likely to overestimate risk for most individuals. In other words, the risk estimates calculated in the BHHRA are likely conservative estimates of actual exposure and adverse effects.

In the BHHRA, numerous assumptions were made regarding the frequency and intensity of potential exposure to contaminated environmental media for individuals within the Study Area. For many of these assumptions, there is no site-specific information or data for activities within the Study Area that may result in exposures to surface water, surface sediment, or ambient air, nor is there information for the recreational consumption of fish and crab tissue. Thus, exposure assumptions were made based on using standard default values or professional judgment, and the uncertainty analysis was conducted to evaluate whether the assumptions used in the BHHRA overestimate or underestimate the actual risks experienced by human receptors in the Study Area.

As presented in the exposure assessment and uncertainty analysis for recreational anglers/crabbers in the BHHRA, there is no site-specific information for recreational fish and crab consumption rates within the Study Area. The BHHRA uses USEPA-recommended fish and crab consumption rates that are based on regional studies and do not account for all the site-specific Study Area conditions that influence the nature of recreational angling that occurs within the Study Area. Several important conditions are the following: the industrial nature of (Newtown Creek is a designated SMIA) and limited public access to the Study Area, which constrain the opportunities for recreational anglers to fish and crab within the Study Area; and the NYSDOH sportfish advisories in place for the East River and tributaries, including Newtown Creek. NYSDOH fish advisories are an exposure control method intended to limit public exposure to chemical contaminants in fish or shellfish that may occur because of contaminated areas like the site itself, while that contamination exists. A BHHRA estimates the current and future baseline risks posed by a site in the absence of exposure control methods, so it was assumed that the RME angler/crabber was not aware of, or had not adhered to, sportfish advisories. However, the portion of the angling population that does adhere to sportfish advisories will have a much lower consumption rate than was assumed in the BHHRA. These conditions directly influence the potential amount of angling and crabbing that occurs within the Study Area, so the use of consumption rates based on regional studies adds additional uncertainty to the potential risks estimated in the BHHRA associated with this exposure pathway.

To evaluate the potential impact of these factors on estimated cancer risks and noncancer hazards, alternative risk estimates were calculated assuming that people fishing and crabbing in the Study Area are in compliance with the NYSDOH advisories (i.e., sportfish advisory scenario) or that only 50% of their consumption of recreationally caught fish and crab comes from the Study Area (i.e., broader recreational scenario; the remaining 50% of their consumption is assumed to have zero contamination). These alternative scenarios were evaluated because they are plausible based on conditions present within the Study Area. The estimated cancer risks and noncancer hazards are lower for these alternative scenarios, but they are still not within the USEPA's acceptable range for cancer risks and are above the noncancer hazard threshold for some receptors. The results of the risk calculations for the broader recreational scenario and sportfish advisory scenarios are presented in Tables 7-9 through 7-11, along with the RME risk and hazard estimates for the BHHRA scenario for each receptor age class and fish/crab species evaluated in the BHHRA. These additional scenarios provide an estimate of potential risks and hazards to recreational anglers/crabbers, considering the characteristics and conditions present in the Study Area.

The estimated Phase 2 reference area RME cancer risks and noncancer hazards are generally consistent with the estimated cancer risks and noncancer hazards as calculated for the alternate broader recreational scenario. Thus, if the Study Area conditions that limit the potential for recreational angling and crabbing within the Study Area (which include access and regulatory restrictions, as well as the presence of alternative high-quality recreational angling access locations in close proximity to the Study Area) were to reduce the proportion of the fish and crab consumed by recreational anglers in the Study Area by 50%, their cancer risks and noncancer hazards would be similar to those estimated for reference area anglers consuming regional fish and crab tissues.

Another key uncertainty associated with the risk assessment is the linkage between risks due to exposure within the Study Area and the ultimate sources of those risks, which is summarized as follows:

- The dominant risk pathway is fish and crab ingestion; body burdens of the fish and crab collected from the Study Area include contaminants accumulated by these species both within and outside the Study Area, based on the life history characteristics of the three species (striped bass, white perch, and blue crab) evaluated in the BHHRA.
- In both the Study Area and the Phase 2 reference areas, cancer risk estimates are either in the upper end of the USEPA acceptable risk range or exceed the acceptable risk range, and noncancer hazards exceed the HI threshold of 1.

The conservative nature of the assumptions used in the BHHRA may lead to an overestimation of risks to human receptors from standard CERCLA COPCs. However, the BHHRA evaluates the potential risks associated with CERCLA hazardous substances only; it does not include an evaluation of potential risks associated with exposure to biological agents such as pathogens. The lack of evaluation of these pollutants and contaminants in the Study Area may underestimate human health risks associated with exposure to surface water and surface sediments within the Study Area.

Additional uncertainties are associated with the chemistry data, exposure assumptions, and toxicities of COPCs. These uncertainties are inherent in any CERCLA risk assessment, and there are standard methods that have been used to ensure that the risk estimates are not underestimated. Taking into account the uncertainties and assumptions, this assessment tends to overestimate risks more than underestimate them, consistent with the health-protective nature of risk assessment. Thus, despite the uncertainties, the baseline characterization of RME risks for the Study Area is considered to be protective of human health (i.e., unlikely to underestimate risk) and sufficient to support risk management decisions. Additional details regarding uncertainty are presented in Section 7 of the BHHRA (see Appendix H).

7.1.7 Baseline Human Health Risk Assessment Conclusions

Sections 7.1.7.1 and 7.1.7.2 present the key conclusions of the Newtown Creek BHHRA.

7.1.7.1 Study Area

The BHHRA evaluated 12 exposure scenarios; of these, potential risks in excess of USEPA's acceptable cancer risk range and noncancer hazard threshold were identified for only the following exposure scenarios:

- RME fish and crab consumption by recreational anglers results in an estimated lifetime excess cancer risk that exceeds the USEPA acceptable cancer risk range of 10^{-4} to 10^{-6} (i.e., up to 8×10^{-4}). The estimated noncancer hazards above the USEPA threshold (HI of 1) are also associated with recreational angler's consumption of fish and crab from Newtown Creek (i.e., up to HI = 40). For all other recreational receptors and pathways, the estimated cancer risks and noncancer hazards are within or below the USEPA acceptable cancer risk range and below the USEPA noncancer hazard threshold.
- The general construction worker is the only occupational receptor with estimated noncancer hazards above the hazard threshold (i.e., HI = 2). The estimated cancer risks for the general construction worker are within USEPA's acceptable risk range.

Furthermore:

- The estimated CTE total cancer risks for the consumption of fish or crab are all within USEPA's acceptable cancer risk range, but the estimated CTE noncancer hazards exceed the HI threshold of 1 for all age classes for the consumption of fish and crab (i.e., up to HI = 5). Total nondioxin-like PCB congeners and TPCB congener TEQ are the COPCs that contribute the highest estimated cancer risks and noncancer hazards for the CTE fish and crabbing scenario.

- For the RME recreational consumption of fish and crab exposure scenario for the Study Area, the primary contributors to both estimated cancer risks and noncancer hazards are PCBs and dioxins/furans (i.e., total nondioxin-like PCB congeners, TPCB congener TEQ, and total dioxin/furan TEQ). For the RME general construction worker, the primary contributor to the estimated noncancer hazards is total nondioxin-like PCB congeners.
- For the CTE scenario for the general construction worker, the estimated cancer risks and noncancer hazards were below USEPA's acceptable risk range and HI threshold of 1.

7.1.7.2 Phase 2 Reference Areas

- The estimated RME cancer risks associated with Phase 2 reference area fish and crab consumption are at the upper end of USEPA's acceptable cancer risk range for striped bass (i.e., up to 1×10^{-4}) and exceed USEPA's acceptable cancer risk range for white perch and blue crab (i.e., up to 2×10^{-4}). The estimated RME noncancer HIs for the reference areas exceed the threshold of 1 (i.e., up to HI = 10).
- These Phase 2 reference area results, along with an understanding of species migration and movement, indicate that fish and crab exposure to COPCs occurs on a regional scale. A portion of the COPCs in the species consumed by recreational anglers in the Study Area may originate from a wide, regional urban area.
- For the RME recreational consumption of fish and crab scenario from the Phase 2 reference areas, the primary contributors to both cancer risks and noncancer hazards are PCBs and dioxins/furans (i.e., total nondioxin-like PCB congeners, TPCB congener TEQ, and total dioxin/furan TEQ).

7.2 Ecological Risk

Section 7.2 provides a summary of the BERA (see Appendix I) conducted for the Study Area. The BERA follows USEPA's process for ecological risk assessment under Superfund (USEPA 1997c) and presents an analysis of risks to aquatic life and semiaquatic wildlife from exposure to hazardous substance releases in the Study Area in the absence of control or mitigation actions. The BERA also takes into consideration the effect of non-CERCLA stressors and other parameters including, but not limited to, DO and habitat quality, which affect the aquatic life and semiaquatic wildlife that use the Study Area.

The BERA uses multiple LOEs to assess risk using chemistry data in surface water, surface sediment, porewater, and tissue of aquatic organisms, as well as surveys of the aquatic life and semiaquatic wildlife that use the Study Area. Specifically for benthic macroinvertebrates, the BERA uses an SQT approach consisting of benthic community analysis, sediment toxicity testing, and both sediment porewater chemistry and bulk sediment chemistry. Bulk sediment chemistry is typically included in an SQT approach, but it does not incorporate consideration of bioavailability; use of measured porewater chemistry and a comparison of sediment AVS to SEM provide data on contaminant bioavailability and, therefore, potential toxicity to benthic invertebrates. The BERA also uses synoptic data and survey information from four Phase 2 reference areas that represent the range of regional impacts for comparison to the Study Area to assess the potential effects from industrial development and discharges from point sources, including MS4s and CSOs (see Section 2.1.2).

7.2.1 Receptors and Exposure Pathways Evaluated

The aquatic organisms and semiaquatic wildlife (the receptors) evaluated in the BERA were selected based on the outcome of surveys conducted in the Study Area in Phase 1 and subsequent discussions with USEPA during development of the *Baseline Ecological Risk Assessment Problem Formulation* (Anchor QEA 2014n).

From the Phase 1 and Phase 2 surveys of the Study Area, the benthic community is dominated by just a few taxa consisting of oligochaetes (segmented worms), polychaetes (bristle worms), and amphipods (small crustaceans; Attachment Bi-A of Appendix Bi and Sections 2.1.4 and 8.3.2 of Appendix I). This is consistent with NYCDEP reports that have described the benthic community of the Study Area as pollution tolerant and exhibiting low species diversity and abundance, especially in the upper reaches of the Study Area (NYCDEP 2011a). Fish and crab community surveys in the Study Area found the dominant fish species to be striped bass, mummichog, and Atlantic menhaden (Attachment Bi-A of Appendix Bi and Section 2.1.4 of Appendix I). The most common species of crab found in the Study Area is the blue crab, followed by the horseshoe crab, which is not a true crab but is a benthic macroinvertebrate from the subphylum *Chelicerata*. Other species of crab found in the Study Area during Phase 1 included the green crab and rock crab (see Phase 1 DSR Submittal No. 1 [Anchor QEA 2013e], Table 3-34). Since the study design for Phase 1 and Phase 2 used sediment grab samples to evaluate the benthic community and submerged crab pots to collect crabs, taxa associated with hard substrates (e.g., rocks, pilings, concrete, and wood bulkheads), such as some mollusks and certain species of crabs, were not targeted and may have been missed. The fish community of the Study Area has been reported to be sparse, especially during the summer months when DO concentrations can drop below 1 mg/L in places (NYCDEP 2011a). This is consistent with low DO found in the Study Area during the Phase 1 and Phase 2 surveys, particularly during the summer months. Bivalves, primarily the ribbed mussel, were found only at a few locations in the Study Area and observed in the crevices of bulkheads and pilings.

The birds and mammals observed during the Phase 1 wildlife surveys are species commonly found in urban environments, such as gulls, rock doves (*Columba livia*), crows, European starlings, Norway rats, feral cats (*Felis catus*), and raccoon (Attachment Bi-A of Appendix Bi). Of the semiaquatic birds, the spotted sandpiper and double-crested cormorant were frequently observed. Others included the black-crowned night heron, green heron, great egret, and infrequently, the belted kingfisher. No amphibians or reptiles were observed in the Study Area during the surveys.

Although aquatic macrophytes were occasionally observed floating in the Study Area near the East River or attached to pilings and riprap during the Phase 1 surveys (Attachment Bi-A of Appendix Bi), none were observed rooted in the Study Area sediment. The only rooted emergent aquatic macrophytes observed were patches of phragmites toward the head of Maspeth Creek. The overall lack of an aquatic macrophyte community is likely due to the physical factors such as characteristics of the substrate, boat and barge traffic, wave action, and slope and light limitations; and it may also be influenced by chemical factors such as elevated porewater sulfide and contaminants that limit their colonization. For example, predominantly vertical bulkheaded shoreline limits the amount of required shallow sloping areas for emergent macrophytes to become established, and the toxicity of contaminants to freshwater and estuarine submerged macrophytes and some estuarine emergent macrophytes has been reported in scientific literature (e.g., Guilizzoni 1991; Lytle and Lytle 2001; Lewis et al. 2001; Yan et al. 2011).

Consistent with the results of the surveys described previously, the receptors and exposure pathways (denoted in parenthesis in the following list) evaluated in the BERA were selected to represent organisms at different ecological trophic levels and with different site use and feeding strategies, and consist of the following:

- Aquatic plants: phytoplankton (surface water)
- Invertebrates: zooplankton, benthic macroinvertebrates, epibenthic invertebrates such as ribbed mussels and blue crab (surface water and surface sediment)
- Fish: striped bass and mummichog (surface water, surface sediment, and diet)[287]
- Semiaquatic birds: spotted sandpiper as representative of invertivorous birds, green heron and black-crowned night heron as representative of invertivorous/piscivorous birds, and double-crested cormorant and belted kingfisher as representative of piscivorous birds (surface water, surface sediment, and diet)
- Semiaquatic mammals: raccoon (surface water, surface sediment, and diet)

In addition to quantifying risks to these receptors, the BERA also includes the following qualitative evaluations:

- Observations of fish and crab presence/absence, richness, and diversity
- Observations of bird and raccoon presence/absence, and for birds, richness and abundance
- Observations of reptile and amphibian presence/absence
- Observations of aquatic macrophyte presence/absence

The relationship between the exposure media and the receptors is presented in Figure 7-3. Given the number of LOEs evaluated in the BERA, overall risks were evaluated using a structured weight-of-evidence (WOE) approach described by USEPA (2016b). The approach identifies those LOEs that are more relevant, stronger, and more reliable than others and have a greater weight in an overall assessment of potential risk. In addition, the WOE evaluation determines whether the LOEs indicate evidence of risk, and (if so) what the

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[287] Atlantic menhaden are included as a prey item to assess risks to striped bass and aquatic-dependent wildlife.

magnitude of that risk might be. The outcome of the BERA WOE is presented as part of the conclusions in Section 7.2.6.

7.2.2 Datasets

The datasets used in the BERA are described in Section 4 of the BERA (see Appendix I). The datasets consist of the following: surface water and surface sediment chemistry; fish, crab, ribbed mussel, and worm (polychaete) tissue chemistry; counts of benthic macroinvertebrates; laboratory-based sediment toxicity tests conducted with benthic macroinvertebrates; paired sediment and tissue chemistry from a laboratory-based bioaccumulation study; fish and crab community surveys; and surveys of wildlife habitat, site use, and feeding behavior. Observations were also made regarding the presence or absence of aquatic plants, amphibians, and reptiles. In addition to the bioaccumulation test, the bioavailability of chemicals to benthic macroinvertebrates was determined by measuring the concentrations of metals and non-polar organic chemicals in sediment porewater during the toxicity tests.

Although bulk sediment chemistry data can be used to understand the relative distribution and concentration of chemicals in Study Area sediment and to evaluate the correlations between bulk sediment concentrations and toxicity, research on equilibrium partitioning demonstrates that porewater is the primary route of exposure to benthic macroinvertebrates for divalent metals and non-polar organic compounds (USEPA 2005b, 2012d, 2017). Therefore, porewater chemical concentrations were determined empirically to provide the most direct measure of exposure to sediment-associated chemicals. The bioavailability of metals was further evaluated by determining the likelihood for some metals to form insoluble sulfides in the sediment by measuring Σ SEM – AVS concentrations in sediment, and by performing sequential extraction to fractionate metals according to their reactivity/bioavailability.

7.2.3 Screening Level Ecological Risk Assessment

To focus the BERA on those contaminants that are likely the most important contributors to ecological risk, a screening level ecological risk assessment (SLERA) was first conducted. The goal of a SLERA is to reduce the number of contaminants and ecological receptors that need to be evaluated further in the baseline risk analyses. To ensure that no contaminants or receptors are erroneously eliminated, a SLERA uses conservative assumptions for exposure and effects. A quantitative SLERA was performed as part of Phase 2 (referred to as the Phase 2 SLERA) for the receptors and exposure pathways discussed in Section 7.2.1 and identified in Figure 7-3, with the exception of the aquatic macrophytes, amphibians, and reptiles, which were evaluated qualitatively in the BERA.

For the Phase 2 SLERA, chemical concentrations in surface water were used to assess potential risk to aquatic life (i.e., phytoplankton, invertebrates, and fish), while chemical concentrations in sediment were also used to assess potential risk to benthic macroinvertebrates. Potential risks were also assessed using a tissue residue approach based on tissue chemical concentrations for benthic macroinvertebrates (represented by polychaetes), bivalves (ribbed mussel), epibenthic decapods (blue crab), and fish (striped bass and mummichog). Potential risks to semiaquatic birds (spotted sandpiper, green heron, black-crowned night heron, belted kingfisher, and double-crested cormorant), and mammals (raccoon), were assessed by evaluating their dietary intake of chemicals (i.e., through the ingestion of prey and incidental ingestion of sediment).

The Phase 2 SLERA used a chemical's FoD, and HQ, to select COPECs. For surface water, surface sediment, and tissue, HQs were calculated by comparing media-specific concentrations to media-specific effect levels or SLs. The surface water and sediment SLs were selected from a USEPA-directed hierarchy of benchmarks (USEPA 2013). The tissue SLs were based on two sets of CBRs—one set was provided by USEPA from the Lower Passaic River Study Area risk assessment (USEPA 2014e) and are referred to as USEPA Region 2 CBRs; the other set is based on effect levels from the USACE Environmental Residue Effects Database (USACE 2013b) and USEPA's PCB Residue Effects Database (USEPA 2007b) and are referred to as the NCG CBRs. For wildlife (birds and the raccoon), HQs were calculated by comparing a TDI to a toxicity reference value (TRV). For sediment, water, and tissue, COPEC selection was based on media-specific, Study Area-wide 95% UCL concentrations; whereas for wildlife, COPEC selection was based on Study Area-wide 95% UCL TDIs.

As a result of the Phase 2 SLERA, the chemicals were placed into one of the following three categories:

- **COPEC**. Chemicals with an FoD greater than 5% and an HQ greater than 1 based on a 95% UCL concentration or 95% UCL TDI were carried forward for further evaluation.
- **Eliminated as a COPEC**. Chemicals with an HQ less than 1 or with an FoD less than 5% were eliminated as COPECs and were not evaluated further.
- **Uncertain COPEC**. Chemicals without an SL or TRV, but with an FoD greater than 5%; or chemicals with an FoD less than 5%, but with an RL greater than their respective SLs, were identified as uncertain COPECs. Uncertain COPECs are discussed in the uncertainty section for each receptor group.

The COPECs by media or receptor are presented in Table 7-12. More COPECs were identified based on bulk sediment chemistry than for any other media or receptors, so these are listed first in Table 7-12. Five COPECs were identified based on surface water chemistry, and thirteen COPECs were identified based on the tissue residue approach when using the USEPA Region 2 CBRs (no COPECs were identified based on the NCG CBRs). As shown in Table 7-12, different tissue-based COPECs were identified for different receptors, ranging from nine for the striped bass to four for bivalves.[288] For wildlife, more COPECs were identified for the spotted sandpiper than for the other birds evaluated or for the raccoon. Three chemicals that were identified as sediment COPECs are uncertain COPECs for surface water, and two are uncertain COPECs based on tissue data (these are shown in Table 7-12). All other uncertain COPECs for surface water, as well as for surface sediment, tissue, and wildlife that are not included in Table 7-12, are tabulated in the BERA (see Appendix I).

7.2.4 Baseline Risk Analyses

For the COPECs identified for further evaluation in the Phase 2 SLERA (see Table 7-12), the baseline risk analyses used site-specific information to reduce uncertainty in the estimates of exposure for the receptors, media, and exposure pathways identified in Figure 7-3. In addition, the baseline risk analyses used alternative effect thresholds for COPECs identified

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[288] Note that although HPAH, TPAH (17), and TPCB congeners are counted as three COPECs, it is acknowledged that each of these COPEC groups consists of multiple individual contaminants.

in the SLERA that were more relevant to the receptors under evaluation. However, it is recognized that each COPEC, as well as other stressors, may also contribute to overall ecological risk. Sections 7.2.4.1 through 7.2.4.5 provide a summary of the baseline risk analyses for aquatic life based on the COPECs identified in surface water, for benthic macroinvertebrates based on the COPECs identified in surface sediment, for fish and invertebrates based on COPECs identified in biota tissue, and for wildlife based on the COPECs identified for the semiaquatic birds and the raccoon. For completeness, the baseline risk analyses for fish included an evaluation based on the dietary intake of PAHs and metals (prey and incidental ingestion of surface sediment) and for benthic fish based on exposure to porewater COPECs.

7.2.4.1 Aquatic Life

Five chemicals were identified as COPECs in the Phase 2 SLERA—Ba, Cu, 2,4′- and 4,4′-DDD, -DDE, -DDT (total DDx), carbon disulfide, and cyanide. Similar to the Phase 2 SLERA, exposures in the baseline risk analyses were based on Study Area-wide 95% UCLs in the baseline risk analyses. In contrast to the Phase 2 SLERA, 95% UCLs used less conservative MDLs for non-detected chemicals, rather than RLs. The use of Study Area-wide 95% UCLs is justified given that the surface water dataset shows little evidence of spatial variation across the Study Area (see Appendix I, Figures 6-1 through 6-5). Study Area-wide surface water 95% UCL concentrations that are used to assess risk are appropriate, because the larger sample size associated with the full dataset improves the precision of the 95% UCLs as conservative estimators of the mean. Any isolated maximum value does not warrant examination on a smaller spatial scale. In the Phase 2 SLERA, surface water SLs were selected from a hierarchy recommended by USEPA. For the baseline risk analyses, alternative threshold values were selected for cyanide, Cu, Ba, and total DDx that are more applicable to the marine species under evaluation. However, in the absence of alternative toxicity data for carbon disulfide, the Phase 2 SLERA SL was used in the baseline risk analyses. Using exposure concentrations based on the MDL for non-detected chemicals (rather than the RL), and using these alternative threshold values, HQs were less than 1.0 for Ba, Cu, total DDx, and carbon disulfide, while the HQ for cyanide was 1.1.

For cyanide, an additional analysis was performed to predict free cyanide levels from total cyanide concentrations. This was necessary because the cyanide SL used in the Phase 2 SLERA is based on free cyanide, rather than the total cyanide measured in the surface water samples. Evaluation of the model-estimated free cyanide concentrations concluded that, of the 360-plus surface water samples that were analyzed (Phase 1 plus Phase 2), there were two samples with higher concentrations than the range of 0.03 to 9.68 µg/L throughout the Study Area. These two samples were collected from Dutch Kills at approximately CM 1.5 (free cyanide concentration of 27.7 µg/L), and English Kills at approximately CM 3.1 (free cyanide concentration of 23.7 µg/L). The Dutch Kills sample was collected from Station DK011 at the head of the tributary, and the English Kills sample was collected from Station EK006, approximately mid-way between the head and mouth of this tributary. If these two sample results are not included in the dataset, the 95% UCL free cyanide concentration is below the chronic threshold value, resulting in an HQ of 0.8 (see Table 7-13).

7.2.4.2 Benthic Macroinvertebrates

Risks to benthic macroinvertebrates were assessed using an SQT approach, which consists of a benthic community analysis, sediment toxicity testing, and bulk sediment chemistry. Rather than rely exclusively on bulk sediment chemistry to assess risks to benthic macroinvertebrates, sediment porewater chemistry was used in conjunction with sediment toxicity test data to provide a more definitive identification of contaminants contributing to benthic macroinvertebrate risk. The assessment also considered other stressors, such as low water column DO and other non-CERCLA stressors, to support interpretation of benthic community composition and effects on benthic macroinvertebrates in the Study Area.

The composition of the benthic community was measured directly by counting taxa in field-collected sediment samples. Taxa were identified to species. Data were evaluated using biological metrics, and Study Area locations were compared to reference conditions. The WBI was selected as the primary metric to evaluate the status of the benthic communities in the Study Area and the Phase 2 reference areas. The WBI was specifically developed by USEPA for estuaries in the northeastern United States (Adams et al. 1998). Since this index is specific to the New York-New Jersey Harbor, it is geographically appropriate to use for the Study Area and the reference areas. Overall, the benthic community is considered stressed, based on Phase 1 and Phase 2 benthic community data. Although none of the sediment COPECs demonstrated a clear relationship with benthic community metrics, DO (a non-CERCLA stressor) did demonstrate a relationship with benthic community metrics spatially and temporally. When measured DO levels were below 3 mg/L (State of New York Class SD saline surface water threshold), benthic community metric scores were lower at some stations at certain times of the year. This was particularly the case for locations in the upper reaches of the Study Area during the summer months, when DO levels are expected to be at their lowest. However, at some stations in the upper reaches of the Study Area, exposure to higher COPEC porewater concentrations may also play a role.

Laboratory-based sediment toxicity tests were completed for 36 Study Area stations and 24 Phase 2 reference area stations, using the amphipod *Leptocheirus plumulosus*, a sensitive benthic macroinvertebrate species. The tests measured 10-day acute survival and 28-day chronic survival, growth (biomass and weight), and reproduction (per surviving amphipod and per surviving female amphipod). Contaminant bioavailability was evaluated by measuring porewater COPEC concentrations, and Σ SEM – AVS was measured in bulk sediment to further help evaluate the bioavailability of Cu, Cd, Pb, Ni, and Zn.

Studies have shown that AVS reacts with the divalent metals, forming insoluble metal sulfides, thereby reducing bioavailability (Ankley et al. 1991, 1996; Berry et al. 1996; Di Toro et al. 1992). Therefore, Σ SEM – AVS was calculated for each of the Study Area and Phase 2 reference area stations, pre- and post-toxicity testing, to provide a measure of metal bioavailability and potential contributions to adverse effects. To further refine the evaluation of metals bioavailability, a subset of sediment samples was selected for sequential extraction (Tessier et al. 1979). Sequential extraction is designed to fractionate metals in a sample according to their reactivity/bioavailability, by subjecting the sample to a sequence of progressively aggressive chemical treatments that target specific chemical forms.

Research and agency guidelines have demonstrated that sediment porewater is the primary route of exposure for benthic macroinvertebrates and a more accurate representation of the bioavailable fraction of the contaminant than bulk sediment chemistry (Arp et al. 2011; Di Toro et al. 1991; Lu et al. 2011; NYSDEC 2014; USEPA 2003c). When coupled with toxicity testing, the relationship between porewater contaminant concentrations and biological responses provides a more definitive identification of contaminants contributing to adverse effects (USEPA 2005b, 2012d, 2017). Therefore, porewater was collected synoptically for TPAH (34), TPCB congeners, and pesticides using SPME fibers, and for metals using mini-peepers. Evaluations to support decision-making include statistical comparisons between the test controls and the Study Area and Phase 2 reference area samples; the development of a reference envelope to establish the magnitude of toxicity relative to expected regional conditions; and the examination of the relationship between toxicity, sediment chemistry, and porewater chemistry. A reference envelope was developed from the results of toxicity tests using the Phase 2 reference area samples. The distribution of reference area endpoint values is defined as the reference envelope, and the 95% LCL on the 5th or 20th percentile of the distribution is defined as the reference envelope value or the threshold defining the reference area population. The sample populations of the four Phase 2 reference areas represent Industrial/Non-Industrial and CSO/Non-CSO conditions; they provide estimates of chemical and ecological conditions that encompass the range of conditions found in the Study Area. Study Area toxicity test results were compared to the reference envelope thresholds using the following reference area datasets:

- A pooled dataset of all four Phase 2 reference areas (n = 48)
- A pooled dataset of all four Phase 2 reference areas with three Westchester Creek stations (n = 6)[289] removed for a total of 42 bioassay samples (n = 42)
- A dataset for Gerritsen Creek (n = 12)
- A dataset for Spring Creek (n = 12)
- A dataset for Head of Bay (n = 12)
- A dataset for Westchester Creek (n = 12)
- A dataset for Westchester Creek with three stations removed (n = 6)

The sediment toxicity tests indicated that 10-day survival and 28-day survival, growth, and reproduction of *Leptocheirus* in CM 2+ and the tributaries—Dutch Kills, Whale Creek, Maspeth Creek, East Branch, and English Kills—are below the Phase 2 reference area thresholds. For example, both the 10-day and the 28-day survival test results show that for stations in these

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[289] There were 6 stations in each of the four Phase 2 reference areas for a total of 24 stations. Sediment toxicity tests for the Study Area were split between two batches, with reference area stations included with each batch, resulting in a total of 48 Phase 2 reference area sample results. Therefore, removal of three stations from Westchester Creek results in the removal of six sample results.

locations, survival is statistically different from the control and lower than the reference envelope thresholds of 63.4% and 59.7% survival, respectively. Conversely, stations in CM 0 – 2 are not statistically different from the controls and exhibit higher survival than the reference envelope threshold for the 28-day test, attaining more than 95% survival at several locations (see Figure 7-4a for the results of the 28-day survival test and Figure 7-4b for the results of the 10-day survival test) but are statistically different from controls and exhibit lower survival than the reference envelope threshold for the 10-day test. Similar patterns to the 28-day survival test results are observed for the 28-day growth (biomass and weight) and reproduction (per surviving amphipod and per surviving female amphipod) toxicity tests. The outcome was similar when using reference envelopes developed for individual reference areas and for the reference envelope developed from a pooled dataset with three stations removed from Westchester Creek. The differences in survival in the 10-day and 28-day tests in specific areas of CM 0 – 2 (i.e., CM 0 – 1) are not unexpected, because the 10-day and 28-day studies represent different exposure conditions, which allows investigators to more fully assess the sediment toxicity. Reasons for these differences are discussed in detail in Section 8.5.3.4.1 of the BERA.

To explore the possible relationships between sediment, porewater, and toxicity, bulk sediment and porewater (where available) concentrations of the sediment COPECs identified during the Phase 2 SLERA were compared to the results of the sediment toxicity tests. The purpose of this comparison is to evaluate whether there are clear concentration-response relationships throughout the range of sediment COPEC concentrations, and if bulk sediment SLs are exceeded, whether porewater COPEC concentrations are also elevated to levels that would drive toxicity. These relationships are presented in a series of x-y plots in the BERA (of which a subset is discussed in this section), which show that although there are clear correlations between many bulk sediment COPECs and toxicity, the multiple exceedances of bulk sediment SLs for metals, PAHs, pesticides, TPCB congeners, and SVOCs make it impossible to determine whether a particular bulk sediment COPEC is contributing more to toxicity than another COPEC. Of the 60 bulk sediment-based COPECs evaluated in the BERA, 44 have porewater concentrations below the surface water-porewater chronic SL, and for many of those 44, porewater concentrations are several orders of magnitude below the SL. The 16 COPECs with porewater concentrations above their respective SL are Cu, Pb, Zn, one parent PAH (benzo(b,k)fluoranthene), and 12 alkylated PAHs.

For Cu, Pb, and Zn, although maximum porewater concentrations exceeded SLs at a few stations in the Turning Basin, Maspeth Creek, and English Kills, bulk sediment Σ SEM – AVS measurements indicated low bioavailability for these metals with average sediment (Σ SEM – AVS) results less than zero (see Figure 7-5). Sequential extraction also supported the lack of metals bioavailability and, therefore, the lack of metals toxicity with respect to bulk sediment exposures, because very few metals were found to be present in an exchangeable form in the sediment samples that were evaluated.

For several PAHs (parent and alkylated), examples of the x-y plots are shown in Figures 7-6 through 7-10. For naphthalene, fluoranthene, and pyrene, Figures 7-6, 7-8, and 7-9, respectively, show that while bulk sediment SLs are exceeded (often by several orders of magnitude), porewater SLs are not. Although porewater concentrations for naphthalene, fluoranthene, and pyrene do not exceed their respective surface water-porewater SL, the porewater concentrations of the alkylated PAHs C4-naphthalenes and C1-fluoranthenes/pyrenes do exceed the SL, as shown in Figures 7-7 and 7-10, respectively. Although these x-y plots show 28-day survival versus porewater concentrations, the results are similar for the 10-day test results.

Furthermore, the BERA demonstrated that there is a high degree of correspondence between the toxic units (TUs) for benzo(b,k)fluoranthene plus the 12 porewater alkylated PAHs and toxicity for stations in English Kills and upper Newtown Creek (i.e., CM 2+) (see Table 7-14). As shown in Table 7-14, the TUs for benzo(b,k)fluoranthene plus the 12 alkylated PAHs for these locations contribute approximately 55% to 100% of the TPAH (34) TU. For many of the samples, the 12 alkylated PAHs contribute more than 90% of the TPAH (34) TU. This supports the finding of USEPA (2003c) with respect to the greater toxicity of the alkylated PAHs when compared to the parent PAHs.

The BERA also identified a subset of stations that had bulk sediment SL exceedances for PAHs and metals, but did not have porewater SL exceedances, yet exhibited both low 10-day and 28-day survival. In other words, the expected COPEC concentration-response pattern was not observed for these stations. Non-COPEC stressors common in urban environments that could have contributed to these findings include elevated levels of porewater sulfide and elevated levels of complex hydrocarbon mixtures. Sulfide is a naturally occurring product of microbial degradation of organic material and is recognized as a confounding factor in bioassay testing. Porewater sulfide concentrations in several 28-day test samples were above levels reported in the literature to impact survival, particularly for stations in East Branch and Maspeth Creek. For example, Station EB006 had very low survival, with a TPAH (34) TU of 1.06—a concentration expected to have only limited effects on *Leptocheirus* survival—but a maximum porewater sulfide concentration of 58 mg/L. Station MC017 had porewater COPEC TUs less than 1 and a maximum sulfide concentration of 16 mg/L, above the 2 mg/L threshold and approaching the 20 mg/L moderate effects benchmark. Of note is that during the 10-day test, the maximum porewater sulfide concentration spiked at 280 mg/L at Station MC017. Because the 10-day acute test is a static, non-renewal test, the overlying water in the test beakers is not replaced. This likely results in a buildup of sulfide and ammonia concentrations and low DO. However, this does not explain all of the toxicity observed in the 10-day test results, because sulfide and ammonia concentrations were not elevated above benchmarks in all areas where survival was low but COPEC sediment and porewater concentrations were not elevated at levels that would result in the observed toxicity. Moreover, 28-day test results in these areas were not reduced below reference area levels.

With respect to complex hydrocarbon mixtures, USEPA guidance on the development of porewater-based sediment remediation goals states the following: "at some sediment sites, PAHs reside in an oily matrix in the sediment, and the oily matrix can contain high levels of aliphatic hydrocarbons (e.g., alkanes and cycloparaffins). Aliphatic hydrocarbons are the major components of lubricants and greases and are present in crude oil and numerous refined petroleum products. A confounding issue with PAHs might occur when high levels of aliphatic hydrocarbons are present in the sediments" (USEPA 2017). The guidance also notes that for filter-feeding benthic invertebrates, the mechanism of toxicity for the aliphatic hydrocarbons "…can stem from a physical effect, such as fouling of respiratory surfaces by the oil phase" (Mount et al. 2015, unpublished results cited in USEPA 2017). Given the urban setting of the Study Area and its documented sediment contamination, complex hydrocarbon mixtures such as those described by USEPA (2017), Scarlett et al. (2007), and Mount (2010) may be a potential confounding factor influencing the results of the toxicity tests in the Study Area. Although not initially identified as sediment COPECs as a result of the screening level risk assessment, further evaluation by USEPA after the BERA was finalized identified certain classes of hydrocarbons, including C19-C36, TPH, and DRO in bulk sediment as potentially contributing to observed toxicity in the 28-day tests (Anchor QEA 2021). As a result of this additional evaluation, USEPA concluded that TPAH (34) and C19-C36 in bulk sediment best represent the observed toxicity to benthic invertebrates, and a discussion of the nature and extent of C19-C36 in the Study Area was subsequently added to Section 4 of this version of the RI Report.

7.2.4.3 Fish

The baseline risk analyses for fish included a tissue residue approach, an assessment of the dietary intake for PAHs and metals, and an evaluation of the porewater exposure pathway for benthic fish.

Using the tissue residue approach and the NCG LOEC-based CBRs, HQs for mummichog from the Study Area are all less than 1 for all COPECs. When using USEPA Region 2 LOECs, HQs for mummichog on a Study Area-wide basis are less than 1 for Pb, Zn, dieldrin, and total dioxin/furan TEQ and are greater than 1 for Cu and TPCB congeners (see Table 7-13). The risk estimates for mummichog from the four Phase 2 reference areas are the same as (or similar to) those for the Study Area, except for TPCB congeners. When using the USEPA Region 2 LOECs, TPCB congener HQs for mummichog are less than 1 for all four Phase 2 reference areas; for the Study Area, the USEPA Region 2 LOEC-based HQ is 9.2. Spatial examination shows that mummichog TPCB congener tissue concentrations exceeded the USEPA Region 2 LOEC in Dutch Kills, the Turning Basin, English Kills, and East Branch, with TPCB congener tissue concentrations in Dutch Kills an order of magnitude higher than the other FSZs.

Using the tissue residue approach and NCG LOECs, HQs for striped bass from the Study Area are all less than 1 for all COPECs. When using USEPA Region 2 LOECs, HQs for striped bass are less than 1 for Cu, Hg, methyl mercury, dieldrin, and DDx and are greater than 1 for 2,3,7,8-TCDD, total dioxin/furan TEQ, and TPCB congeners (see Table 7-13). The risk estimates for striped bass from the four Phase 2 reference areas are the same as (or similar to) those for the Study Area, except for TPCB congeners. When using the USEPA Region 2 LOEC, the TPCB congener HQ for striped bass from the four Phase 2 reference areas ranges from 0.79 for Head of Bay to 1.5 for Westchester Creek; for the Study Area, the USEPA Region 2 LOEC-based HQ is 4. It is believed that the distances traveled by this migratory species and its movement in and out of the Study Area contribute to the TPCB congener tissue concentrations. The movement of striped bass in and out of the Study Area, and therefore, the contributions of Study Area and regional COPEC exposure, is an important consideration for remedial decision-making.

For dietary intake, TDIs were calculated using an exposure model of the fish diet for striped bass and mummichog, as well as measured concentrations of PAHs and metals in prey and surface sediment. TDIs were compared to fish dose-based TRVs. Striped bass were evaluated on a Study Area-wide basis, due to their wide-ranging foraging activities; mummichog were evaluated on a subarea basis, reflecting their small home range relative to the size of the Study Area. No risks were identified for striped bass using the entire Study Area, and no risks were identified for the mummichog in CM 0 – 2. For mummichog in CM 2+, dietary intake resulted in an HQ of 1.2 for Cu (see Table 7-13).

Evaluation of the porewater exposure pathway for fish is only applicable to benthic fish, such as the mummichog assessed in the BERA. The threshold values used to evaluate benthic fish are the same values used to evaluate porewater exposure of benthic macroinvertebrates, with the exception of TPCB congeners. Invertebrates are not as sensitive to TPCB congeners as fish, because they do not have the aryl hydrocarbon receptor and because they metabolize TPCB congeners differently (Fuchsman et al. 2006). Therefore, the threshold value used for TPCB congeners was based on a value selected by USEPA from early life stage tests with the marine sheepshead minnow (USEPA 1980). Using these threshold values, porewater TUs ranged from 0.46 to 270 for TPAH (34) and from 0.15 to 7.2 for SEM metals, with TUs exceeding 1 at stations in CM 2+ and the tributaries (see Table 7-13). Porewater HQs for TPCB congeners ranged from 0.052 to 9.4, with HQs exceeding 1 at three stations in CM 2+ and four stations in English Kills (see Table 7-13).

7.2.4.4 Polychaetes, Bivalves, and Blue Crab

The baseline risk analyses for polychaetes, bivalves, and blue crab were based on the tissue residue approach.

For polychaetes used in the bioaccumulation study, all HQs were less than 1 when using the NCG LOECs. When using the USEPA Region 2 LOECs, Study Area-wide HQs were 1.0 for HPAH, 1.2 for TPAH (17), and 15 for TPCB congeners (see Table 7-13). The tissue concentrations of these COPECs showed a tendency to increase moving upstream, with English Kills and the Turning Basin exhibiting tissue concentrations above the USEPA Region 2 LOEC values for total HPAH, TPAH (17), and TPCB congeners. Although dieldrin was not identified as a COPEC on a Study Area-wide basis, because the HQ for dieldrin was less than 1 when incorporating tissue data throughout the Study Area, tissue concentrations exceeded the USEPA Region 2 LOEC for sediment collected from one of two stations in English Kills, indicative of exposure to elevated dieldrin levels in this localized area.

For the caged bivalves, all HQs were less than 1 when using the NCG LOECs. When using USEPA Region 2 LOECs, Study Area-wide HQs were 1.9 for TPAH (17) and 3.9 for TPCB congeners (see Table 7-13). The tissue concentrations of these COPECs showed a tendency to increase moving upstream, with Maspeth Creek and English Kills exhibiting tissue concentrations above the USEPA Region 2 CBR LOEC values for total HPAH, TPAH (17), and TPCB congeners. Bivalve tissue concentrations of TPCB congeners exceeded the USEPA Region 2 CBR LOEC values at all but the most downstream Newtown Creek location and in Whale Creek. Although dieldrin was not identified as a COPEC on a Study Area-wide basis for the same reason as discussed above for polychaetes, and was only detected in caged bivalves from English Kills and Maspeth Creek, tissue concentrations at both locations exceeded the USEPA Region 2 LOEC, indicative of exposure to elevated dieldrin levels in these localized areas.

For blue crab, all HQs were less than 1 when using the NCG LOECs. When using USEPA Region 2 LOECs, Study Area-wide HQs were 1.6 for Cu and 8.8 for TPCB congeners (see Table 7-13). Neither Cu nor TPCB congeners showed a spatial trend in the Study Area; however, TPCB congener concentrations were higher for blue crab collected toward the head of Dutch Kills. For comparison with the Study Area, risk estimates were calculated for blue crab collected from the four Phase 2 reference areas. When using USEPA Region 2 LOECs, the HQs for Cu and TPCB congeners were 1.5 and 4.9, respectively, in Westchester Creek—the Phase 2 reference area in the Industrial/CSO category and most similar to the Study Area.

7.2.4.5 Wildlife

For the baseline risk analyses, wildlife exposure models were developed incorporating site-specific data collected during the Phase 2 wildlife surveys on Study Area use, foraging activity, and prey type (see Section 7.2.4.6.2). For example, the green heron and black-crowned night heron (as well as other birds in the same feeding guild) were mostly observed feeding on fish, followed by invertebrates other than crab. Therefore, the dietary proportions for these birds were modified to 90% fish and 10% polychaetes, rather than the 50% fish, 25% blue crab, and 25% polychaetes used in the Phase 2 SLERA. In addition, during the Phase 2 wildlife surveys, the double-crested cormorant was observed flying between the Study Area and a colony on U Thant Island in the East River, and also north and south of the island along the East River, past the Study Area. Based on these observations, the double-crested cormorant was evaluated using a range of exposure-modifying factors from 0.25 to 1.0 to reflect that these birds could spend significant time foraging outside the Study Area, in other regions of New York Harbor, or that these birds could spend all their time foraging within the Study Area. Exposure modifying factors in the baseline risk analyses also accounted for the seasonal migration of birds, the availability of soft substrate for probing birds such as the spotted sandpiper, and the reduced foraging activity of raccoon in aquatic habitats relative to preferred foraging near human use areas.

Based on site-specific exposure assumptions, HQs for the double-crested cormorant and the raccoon were all less than 1. Using a range of exposure modifying factors for the double-crested cormorant from 0.25 to 1.0, TPCB HQs ranged from 0.26 to 1.03. For the spotted sandpiper, HQs exceeded 1 for Pb (HQ = 1.6) and for TPCB (HQ = 1.7). For the green heron, black-crowned night heron, and belted kingfisher, HQs exceeded 1 for TPCB (HQs of 2.3, 1.7, and 1.8, respectively) (see Table 7-13). The areas contributing most to the dietary exceedances are Dutch Kills, Maspeth Creek, and English Kills for Pb in sediment and Dutch Kills for TPCB in mummichog as a prey item.

7.2.4.6 Biological Surveys

In addition to quantifying risk from exposure to contaminants, the BERA also included surveys of fish and crab in the Study Area and the Phase 2 reference areas, and surveys of the habitat and wildlife (birds and mammals) in the Study Area and the Phase 2 reference areas.

7.2.4.6.1 Fish and Crab Surveys

Fish and crab surveys were conducted in the Study Area and the Phase 2 reference areas for the following two reasons:

- To collect fish and crab for tissue analysis as part of the quantitative baseline risk analyses
- To provide data for a qualitative assessment of the distribution of fish and crab within the Study Area and the Phase 2 reference areas

Fish and crab were collected from six FSZs in the Study Area (FSZs 1, 2, 3, 4a, 4b, and 5) and in one FSZ in each of the Phase 2 reference areas, over two sampling events (see Section 2.1.5.4). The zones were defined in this way to ensure that there was an attempt to collect fish and crab samples in a number of locations within the Study Area, rather than a focus on only one or two segments. The first sampling event took place in the spring from June 2 to July 2, 2014, and the second took place in the summer from August 4 to September 5, 2014.

The most common species of fish captured in the Study Area and the Phase 2 reference areas were mummichog, Atlantic menhaden, and striped bass. Atlantic silverside (*Menidia menidia*) were also common in Spring Creek and Gerritsen Creek but not in the Study Area. A number of fish species, such as the northern kingfish (*Menticirrhus saxatilis*), tautog (*Tautoga onitis*), sea robin (*Prionotus carolinus*), and spiny dogfish (*Squalus acanthias*), were documented in the Phase 2 reference areas but were absent from the Study Area. Five species of crab were found, of which four were found only in the Phase 2 reference areas. Blue crab were found in the Study Area and the Phase 2 reference areas.

Differences in community composition between the Study Area and the Phase 2 reference areas were assessed using species richness and the Shannon index of species diversity (Spellerberg and Fedor 2003), which takes into account species richness and the relative abundance of different species.

Fish and crab species richness and diversity in the Study Area show a general trend of higher richness and diversity in the main stem of the creek, with decreasing richness and diversity in CM 2+ and the tributaries. Reference area species richness and diversity are highest in Gerritsen Creek (the Jamaica Bay reference area in the Non-Industrial/Non-CSO category), with decreasing richness and diversity in Head of Bay and Spring Creek, and lowest richness and diversity in Westchester Creek (Industrial/CSO category). Westchester Creek is the Phase 2 reference area with attributes most similar to the Study Area. In general, species richness and diversity is higher in the three Phase 2 Jamaica Bay reference areas (i.e., Head of Bay, Gerritsen Creek, and Spring Creek) than the Study Area. Furthermore, species richness and diversity of Westchester Creek are closer to that of the Study Area than the other three Phase 2 Jamaica Bay reference areas.

7.2.4.6.2 Wildlife Surveys

Wildlife surveys were conducted in the Study Area and the Phase 2 reference areas in the spring (May 19 through June 27, 2014), and summer (August 4 through September 12, 2014). The surveys were performed in two parts—a general survey and a species-specific survey. A general survey was conducted to identify suitable habitat features for, and the presence of, the target species—spotted sandpiper, green heron, black-crowned night heron, belted kingfisher, double-crested cormorant, and raccoon. Using the results of the general survey, a species-specific survey was conducted to collect data on site use, foraging activity, and prey type, where possible. The species-specific surveys consisted of repeated observations from established survey points identified during the general surveys. The surveys were conducted at various times between sunrise and sunset to capture target species and their behavior over different times of day (see Section 11.1.1.1 of Appendix I).

Although there are a number of different species of aquatic-dependent birds that use the Study Area and the Phase 2 reference areas, species richness in the Study Area is lower than in the Phase 2 reference areas. For the Study Area, the number of species ranged from 9 to 10 for spring and summer, respectively. In contrast, for the Jamaica Bay reference areas, the number of species ranged from 20 in the spring to 24 in the summer. Within the target feeding guilds, there are also some notable differences between the Study Area and the Phase 2 reference areas. For the invertivorous feeding guild (represented by the spotted sandpiper), and the invertivorous/piscivorous feeding guild (represented by the green heron and the black-crowned night heron), although the numbers of birds observed in the

Study Area and the Phase 2 reference areas were similar, there were more species observed within these feeding guilds in the Phase 2 reference areas. For the piscivorous birds, both double-crested cormorants and belted kingfishers were observed in the Study Area and the Phase 2 reference areas. The double-crested cormorant was observed throughout the Study Area and in the Phase 2 reference areas in the spring and summer. As previously noted, the double-crested cormorant could spend significant time foraging outside the Study Area, because it was observed flying between the Study Area and U Thant Island in the East River, as well as flying north and south of the island along the East River, past the Study Area. Only one belted kingfisher was observed in the Study Area in the spring, and only two in the summer. No belted kingfishers were observed in the Phase 2 reference areas in the spring, but five were observed in the summer. The only mammals observed during the surveys were Norway rats, feral cats, and a few raccoons. No amphibians were observed in the Study Area or the Phase 2 reference areas and no reptiles were observed in the Study Area. Snapping turtles (*Chelydra serpentine*) were often observed on the banks of Head of Bay, Spring Creek, and Gerritsen Creek, and the diamondback terrapin (*Malaclemys terrapin*) was occasionally observed in Spring Creek and Head of Bay.

The habitat surveys indicate differences between the Study Area and the Phase 2 reference areas related to several factors, including the following:

- The Jamaica Bay reference areas (Head of Bay, Spring Creek, and Gerritsen Creek) exhibit a greater percentage of vegetated shoreline than the Study Area, provide habitat for birds perching and diving in search of prey, and provide cover for wildlife foraging along the shoreline (see Figure 7-11).
- Much of the Study Area shoreline comprises vertical bulkhead with limited intertidal areas for wildlife foraging and limited soft substrate preferred by invertivorous feeding birds.
- As previously discussed (see Section 7.2.4.6.1), the results of the fish and crab surveys demonstrate that the Jamaica Bay reference areas provide a greater diversity of potential prey species for the invertivorous/piscivorous and piscivorous feeding guilds, as well for the raccoon, than does the Study Area.
- Habitat limitations, such as bulkheading and sparse shoreline vegetation, are likely responsible for the lack of amphibians and reptiles found in, or associated with, the shoreline of the Study Area.

- The lack of rooted submerged aquatic macrophytes in the Study Area is attributed to a combination of physical and chemical factors that limit the colonization of aquatic macrophytes. Factors such as boat and barge traffic, wave action, slope limitations of the shoreline, light limitations, potentially elevated porewater sulfide, and COPEC concentrations likely contribute to the observations of very little aquatic macrophyte vegetation in the Study Area.

7.2.5 Uncertainty Analysis

As with the BHHRA, uncertainty is inherent in the BERA and is related to whether the risk analyses accurately represent the potential risks to the ecological receptors exposed to contaminants in various Study Area environmental media. Although the baseline risk analyses used site-specific information and measured COPEC concentrations in exposure media to assess bioavailability, there is uncertainty associated with deriving exposure concentrations or dietary intake. Similarly, although the baseline risk analyses selected effect thresholds to be protective of the receptors, the lack of species-specific data or applicable data for some adds to the uncertainty of the risk calculations. This section summarizes these uncertainties and evaluates whether they could result in an overestimation or underestimation of risk.

With regard to uncertainties related to exposure, the surface water aquatic life Phase 2 SLERA and baseline risk analyses used Study Area-wide 95% UCL contaminant concentrations. Using a Study Area-wide concentration does not provide an evaluation of smaller-scale variations in contaminant concentrations within the Study Area. This could overestimate or underestimate risk if there were small-scale variations in contaminant concentrations. However, an examination of the spatial distribution in the surface water COPECs in the Study Area did not reveal any variations that warranted a subarea evaluation. Furthermore, using 95% UCL concentrations is a conservative estimate of the mean contaminant concentrations, so it may overestimate risks to aquatic life in the Study Area.

In the Phase 2 SLERA, bulk sediment chemistry and generic bulk sediment SLs based on a hierarchy provided by USEPA were used to identify sediment-based COPECs. Because bulk sediment chemistry concentrations are limited in their ability to predict bioavailability and toxicity, use of these data could overestimate or underestimate risk. In the baseline risk analyses, toxicity test data, measured porewater chemical concentrations, Σ SEM – AVS data, and sequential extraction data were used to reduce these uncertainties in the benthic macroinvertebrate risk assessment.

The Phase 2 SLERA tissue residue approach for fish used Study Area-wide 95% UCL tissue concentrations, which could overestimate or underestimate risk if there were small-scale variations in contaminant concentrations. However, for migratory species, such as striped bass, that travel and forage over large distances, a Study Area-wide evaluation is appropriate. In addition, using 95% UCL tissue concentrations is a conservative estimate of the mean contaminant concentrations in the tissue, so it may overestimate risk to the fish. There are also uncertainties with the bivalve and polychaete tissue data used in the Phase 2 SLERA tissue residue approach, because bivalve tissue was collected from ribbed mussels as part of a caged bivalve study, and polychaete tissue was collected as part of a laboratory-based bioaccumulation test. Differences in exposure due to a caged environment or a laboratory setting, rather than exposure in the Study Area, are unknown. These could result in an overestimation or underestimation of risk. However, for both these receptors, 95% UCL tissue concentrations were used as a conservative estimate of the mean contaminant concentrations in the tissue and may overestimate risk.

For wildlife, while site-specific exposure data were available for some of the parameters used in the exposure models, for other parameters, data were obtained from the literature or were based on best professional judgment. Using these could result in an overestimation or underestimation of risk. The use of the lowest gender body weights likely overestimated risk. Since fish body burdens are known to increase with size, risks for the belted kingfisher were likely overestimated, because the only Atlantic menhaden (prey of the belted kingfisher) collected were greater than the target size of 150 millimeters. For those receptors with polychaetes in their diet (spotted sandpiper, green heron, and black-crowned night heron), the use of polychaete tissue data from a laboratory-based bioaccumulation study could result in an overestimation or underestimation of risk.

With regard to the effect levels, the SL hierarchy used in the surface water aquatic life Phase 2 SLERA was provided by USEPA. Some of the SLs were less relevant because they were based on protecting higher trophic-level organisms through food-chain exposure or because they were freshwater-based. Use of these SLs in the Phase 2 SLERA could result in an overestimate or underestimate of risk. For others, more recent toxicity data are available. Further evaluation of the SLs used to identify the COPECs (Ba, Cu, cyanide, and DDx) suggests that these SLs were conservative, so they likely overestimated risk. Alternative chronic threshold values were used for the baseline analyses that are more relevant to the marine species under evaluation.

For the tissue residue approach, two sets of CBRs were used to assess potential risks for polychaetes, bivalves, fish, and crab—NCG CBRs and USEPA Region 2 CBRs. Both sets of CBRs have uncertainties associated with them. The NCG CBRs were selected from the USACE Environmental Residue Effects Database (USACE 2013b) and from USEPA's PCB Residue Database System (USEPA 2007b), rather than from primary literature sources. Although these databases do contain species-specific data, if adequate species-specific information is not available, SLs derived from a suitable combination of studies and species were used. Species vary with respect to their ability to uptake, metabolize, and depurate chemicals, so species-to-species extrapolation could introduce some uncertainties that might result in an overestimation or underestimation of potential risk. Some of the USEPA Region 2 CBRs were not based on whole-body tissue residues, whereas others used alternative measurement endpoints rather than survival, growth, or reproduction. For example, an oyster TPCB CBR used to evaluate polychaetes, bivalves, and blue crab was estimated by applying the ratio of lipids in adult oyster tissue to eggs to oyster egg threshold concentrations from dietary exposure studies; an Atlantic salmon TPCB CBR used to evaluate mummichog and striped bass was derived from a behavioral study with smolt that were aqueously exposed as eggs. Because the measure of effect for the study was sublethal alterations in smolt physiology and behavior, the no observed effect concentration and LOEC derived from this study were exceptionally protective of fish that use the Study Area.

The BERA used two toxicity studies with different exposure conditions—a 10-day exposure with no feeding or water renewal and a 28-day exposure with exogenous feeding and daily renewal of overlying test water. The 10-day study was an acute (short-term) exposure designed to assess survival of organisms that only have the test sediment to process and obtain organic matter to eat. Depending on the OC content of the sediment and the composition of the organic content (i.e., whether the organic matter is processable as food for the organisms or organic chemical contamination), the exposed organisms may demonstrate stress (e.g., mortality) that might not be apparent in a test with exogenous (laboratory-prepared) food. Excluding feeding and water renewal procedures has been shown to impact the health of the organisms and performance of the test (McGee et al. 1993, 2004). The 28-day study was a chronic (long-term) exposure designed to assess survival, along with sublethal effects (growth and reproduction). The 28-day test period is too long for organisms to survive without exogenous food, so laboratory-prepared food was added to each test chamber daily, along with the renewal of overlying test water. Test water is renewed in toxicity tests that are fed to prevent the buildup of food and metabolic wastes, which can negatively impact water quality and skew test results. The 28-day test results provide an assessment of long-term contaminant exposures. USEPA guidance acknowledges that chronic tests are more toxicologically relevant, have greater resolution than acute tests, and are more appropriate for organisms that spend most of their time on site (USEPA 1994, 2014c). However, the 10-day acute study was included in the BERA to provide information on the different exposure conditions, and it allows investigators to more fully assess the sediment toxicity. For both the 10-day and the 28-day tests, the number of triad stations in the Study Area and the Phase 2 reference areas allowed for the development of a reference envelope and an analysis of Study Area toxicity on a small spatial scale, thereby limiting the effect of any uncertainty in the results. The 28-day toxicity test results can be explained based on porewater COPEC concentrations for a majority of the stations sampled in the Study Area, although there are some stations for which the relationship between porewater COPEC concentrations and toxicity appears to be confounded and, therefore, uncertain. Although there are clear correlations between many bulk sediment COPECs and toxicity, multiple exceedances of bulk sediment SLs make it impossible to determine whether a particular sediment COPEC is actually contributing to toxicity. In addition, for many sediment COPECs, high toxicity is observed over a wide concentration range for which a concentration-response relationship is not readily apparent. For these reasons, non-COPEC stressors may play a role in explaining toxicity, which is a source of uncertainty.

For the fish dietary assessment, only a few dietary-based measures of effect are available for fish. In some instances, only one reliable no observed adverse effect level or lowest observed adverse effect level could be found. Therefore, for those bioaccumulative metals and PAHs for which dietary effect levels are not available, risks are uncertain. For wildlife, the dose-based TRVs used are typically derived from laboratory-based studies using test species to represent those in the natural environment. The TRVs could result in an underestimation or overestimation of risk. To minimize underestimating risks, the lowest observed adverse effect level data are typically selected to derive the TRVs. For some of the contaminants, risks are uncertain due to the lack of an SL or dose-based TRV.

7.2.6 Baseline Ecological Risk Assessment Conclusions

The BERA used multiple LOEs to assess risk from exposure to CERCLA COPECs to aquatic life and semiaquatic wildlife that use the Study Area. Quantitative risk analyses were performed for a combination of media and receptors consisting of surface water, surface sediment, and tissue (polychaetes, caged bivalves, blue crab, and fish), and benthic macroinvertebrates, fish, semiaquatic birds, and raccoon based on dietary intake. These analyses used site-specific chemical, biological, and survey data; biological testing; and measures of bioavailability in the Study Area and in the Phase 2 reference areas. Qualitative evaluations consisted of observations for fish and crab presence/absence, richness, and diversity; wildlife presence/absence; and bird richness and abundance, as well as observations for the presence/absence of amphibians, reptiles, and aquatic macrophytes.

The LOEs were used in an overall WOE to inform which LOEs should have a greater role in making risk management decisions for the Study Area (USEPA 2016b). For the Newtown Creek BERA, USEPA recommended the use of a WOE framework and scoring system developed by Menzie et al. (1996). The WOE framework developed by Menzie et al. (1996) involves consideration of 10 specific attributes for each LOE to determine how well the LOE represents the assessment endpoint. The 10 attributes are assigned scores based on the following: 1) strength of association between assessment endpoints and measurement endpoints; 2) data quality; and 3) study design and execution. The resulting scores are presented in Table 7-15 for each LOE. Most of the LOEs received a score of 4 or 5, reflecting the strong association between the assessment and measurement endpoints, the high quality of the site-specific data used in the BERA, and a thorough study design that was successfully executed. The quantitative LOEs were used with the results of the risk analyses described in the preceding sections and summarized in Table 7-13, in an overall evaluation for the evidence of harm and magnitude of response as shown in the final two columns of

Table 7-15. The outcomes are either no evidence of harm or evidence of harm for which the magnitude of response can be either low or high.

The surface water LOE shows there is evidence of harm, but the magnitude of the response is considered low because all HQs were less than 1.0 with the exception of cyanide. For cyanide, the HQs ranged from 0.8 to 1.1, depending on whether two of the 360-plus surface water samples with cyanide concentrations above the Study Area range were, or were not, included in the risk calculations. Other LOEs for which there is evidence of harm, but the magnitude of the response is considered low, are as follows: the community metrics used to evaluate benthic macroinvertebrates, the tissue residue approach using USEPA Region 2 CBRs for bivalves and mummichog, and the dietary LOEs for fish and birds. The benthic community metrics demonstrated that the benthic community in the Study Area is stressed and generally similar to the four Phase 2 reference areas. None of the sediment COPECs demonstrated a clear relationship with benthic community metrics. The tissue residue approach showed that for bivalves, Study Area-wide TPAH (17) and TPCB congener HQs were 1.9 and 3.9, respectively, and for mummichog the Cu HQ was 2.1. Although the Study Area-wide, tissue-based TPCB congener HQ for mummichog was 9.2, as described in Section 7.2.4.3, this was driven by tissue concentrations for mummichog collected from the head of Dutch Kills, which were an order of magnitude higher than elsewhere. For the caged bivalves, tissue concentrations of TPAH (17) and TPCB congeners showed a tendency to increase moving upstream, with the highest concentrations in caged bivalves from English Kills, followed by Maspeth Creek. There is no evidence of harm to bivalves and mummichog when NCG CBRs were used in the tissue residue approach. For wildlife evaluated using the dietary LOE, Study Area-wide HQs for TPCB congeners ranged from 1.7 for the spotted sandpiper and the black-crowned night heron to 2.3 for the green heron. These HQs were driven by dietary intake of mummichog from Dutch Kills. For the spotted sandpiper, Study Area-wide HQs for Cu and Pb were 1.04 and 1.6, respectively, primarily driven by sediment concentrations in Maspeth Creek. There is no evidence of harm for the double-crested cormorant or the raccoon based on the dietary LOE.

LOEs for which there is evidence of harm and for which the magnitude of response is considered high are the following: the tissue residue approach using USEPA Region 2 CBRs for polychaetes, crab, and striped bass; the sediment, porewater, and toxicity test LOEs used to evaluate benthic macroinvertebrates; and the porewater LOE for benthic fish. As for bivalves and mummichog, the outcome of the tissue residue approach for polychaetes, blue crab, and striped bass is uncertain, because there is no evidence of harm when using the NCG CBRs. However, there are some spatial patterns in tissue concentrations worth noting. For blue crab, TPCB congener tissue concentrations show no consistent spatial trend, but they are higher in blue crab collected toward the head of Dutch Kills. For striped bass, there is a general trend for TPCB congener tissue concentrations to increase upstream, but there is also a wide range in concentrations within any one segment of the Study Area. The distances traveled by this migratory species and its movement in and out of the Study Area are believed to contribute to the TPCB congener tissue concentrations (see discussion in Section 6.6.2.2). The movement of striped bass in and out of the Study Area and the contributions of Study Area and regional COPEC exposure are important considerations for remedial decision-making. For polychaetes, there is a trend of increasing TPAH (17) and TPCB congener tissue concentrations in the upstream direction, reflecting the general trend for increasing bulk sediment concentrations moving in the upstream direction. This trend is also generally reflected in the results of laboratory-based acute and chronic sediment toxicity tests conducted over 10 days and 28 days, respectively. The 28-day tests showed that survival, growth, and reproduction of the amphipod *Leptocheirus plumulosus* for samples collected in CM 0 – 2 were above (less toxic than) a threshold based on the pooled Phase 2 reference area data. Conversely, the results of the 10-day tests showed that survival was more toxic than the Phase 2 reference area data. In contrast, for samples in CM 2+ and the tributaries, the results of both the 10-day and 28-day tests were below (more toxic than) the pooled Phase 2 reference area threshold.

As discussed in Section 7.2.4.2, although bulk sediment concentrations for multiple COPECs were correlated with toxicity, it was not possible to attribute toxicity to exposure to any specific COPEC. In contrast, when porewater data were used, a relationship between increased toxicity and increased concentrations of COPECs was evident. Although the bulk sediment concentrations for many of the COPECs exceeded sediment benchmarks, most of those COPECs had porewater concentrations below their surface water-porewater chronic SL, often several orders of magnitude below the SL. The COPECs with porewater concentrations above surface water-porewater chronic SLs were PAHs (mostly alkylated PAHs), with minimal contribution from metals at a few locations. The highest exceedances for porewater PAHs were for locations in CM 2+ and the tributaries, where the magnitude of response based on the results of the 28-day toxicity tests was high. As discussed in the BERA, although toxicity in these locations could be attributed to porewater PAHs, non-COPEC stressors such as low DO, high porewater sulfide, and bulk sediment concentrations of complex hydrocarbon mixtures may have also contributed to the adverse effects to *Leptocheirus* observed in samples collected from some locations. Subsequent to the completion of the BERA, further evaluation by USEPA concluded that bulk sediment concentrations of TPAH (34) and C19-C36 best represented toxicity to benthic invertebrates.

Based on the WOE evaluation completed in the BERA, there is evidence of harm to a number of receptors in the Study Area from exposure to multiple stressors. The magnitude of response is highest in CM 2+ and the tributaries—primarily from exposure to PAHs for benthic macroinvertebrates, bivalves, and benthic fish (mummichog)—to PCBs for benthic macroinvertebrates, bivalves, mummichog, spotted sandpiper, green heron, black-crowned night heron, and belted kingfisher; there is some contribution from Cu and Pb for benthic macroinvertebrates and the spotted sandpiper. Although there is some evidence of harm in CM 0 – 2, based on some LOEs (tissue residue analysis for blue crab, striped bass, and mummichog based on USEPA Region 2 thresholds, and the 10-day sediment toxicity test for benthic macroinvertebrates), the overall WOE evaluation indicates that the magnitude of response is lower in this segment of the Study Area compared to CM 2+ and is often comparable to the magnitude of response in the Phase 2 reference areas. For example, the TPCB congener HQ for blue crab collected from Westchester Creek was 4.9, which is comparable to the HQ of 6.6 for blue crab from Study Area CM 0 – 2 (FSZ 1 and 3); and the dioxin/furan TEQ HQ for striped bass from Head of Bay of 3.0 is comparable to the HQ of 2.7 for striped bass from CM 0 – 2. For Cu, HQs for mummichog from Head of Bay and Gerritsen Creek of 2.8 and 2.9, respectively, are comparable to the HQ of 2.3 for mummichog from CM 0 – 2; and for blue crab, the HQ of 1.5 from Westchester Creek is comparable to the HQ of 1.6 from CM 0 – 2.

In addition, as noted previously, the BERA also demonstrated that the risk analysis results for benthic macroinvertebrates were confounded by non-CERCLA stressors. Surface water DO concentrations below 3 mg/L contribute non-CERCLA-related stress, particularly in CM 2+ and the tributaries during the summer months. Non-COPEC stressors such as elevated porewater sulfide and bulk sediment concentrations of complex hydrocarbon mixtures may also contribute to the adverse effects for benthic macroinvertebrates in locations where these stressors are elevated. Therefore, when using the results of the BERA in the Newtown Creek RI, it is important to consider the influence of these other stressors when making any risk management decisions.

8 CONCEPTUAL SITE MODEL

8.1 Introduction

According to USEPA contaminated sediment guidance (USEPA 2005a) and NYSDEC (2010b), a CSM generally is a representation of the environmental system and the physical, chemical, and biological processes that determine the transport of contaminants from sources to receptors. As such, the CSM provides the current understanding of processes affecting the Study Area and gets updated throughout the RI/FS process as new information becomes available. The CSM is one of many tools that will be used to inform the remedy selected for the Study Area.

A CSM describes the following:

- Sources of potentially significant loads of contaminants
- Nature and extent of contaminants
- Important fate and transport characteristics, which include both chemical-specific (e.g., sorption) and site-specific characteristics (e.g., hydrodynamics)
- Potential exposure pathways
- Potentially impacted receptors

Ongoing external inputs of contaminants to the Study Area originate from point sources (CSOs, MS4s, WWTP-treated effluent, permitted industrial discharges, other permitted/non-permitted discharges), overland flow/direct drainage, tidal exchange with the East River, groundwater discharge to the base of the Study Area, atmospheric deposition, shoreline seeps including NAPL seeps, lateral groundwater discharge from upland properties, bank erosion, and overwater activities. These ongoing external inputs are shown in Figure 8-1. The key inputs (based on flow volumes and contaminant load calculations discussed in the remainder of this section) are point sources and overland flow/direct drainage, the East River, and groundwater discharge to the base of the Study Area (lateral groundwater discharge will be evaluated more fully as part of the CFT modeling effort and the planned study by the USEPA to collect empirical data to reduce uncertainty in the current estimates). The base of the Study Area is defined as the interface between sediment and native material, as well as between sediment and fill (see Figure 8-1 and Section 8.5.2.3). Figure 8-1 also presents fate and transport processes—deposition, resuspension, diffusive and tidal exchange fluxes from porewater, partitioning, and gas ebullition (see Sections 6 and 8.6).

The human and ecological receptors for which risks were determined to exceed USEPA risk thresholds (HIs and HQs greater than 1, cancer risks greater than 1 x 10^{-4}) are shown in Figure 8-2, with food web exposure pathways indicated (see Sections 7 and 8.7 and Appendices H and I for more detail). Human receptors include recreational anglers and crabbers exposed to COPCs through fish and crab consumption and construction workers exposed (to a lesser degree) to COPCs via direct contact with surface sediment. Ecological receptors include benthic macroinvertebrates exposed to COPECs in bulk sediment and porewater; as well as fish, crab, and semiaquatic birds exposed to sediment COPECs directly through incidental ingestion of sediment or indirectly through food web exposure pathways.

A key finding of this RI is that the reaches of the Study Area (CM 0 – 2, CM 2+, and each tributary) differ materially in contaminant distributions, contaminant sources, relative contributions of historical versus ongoing sources, fate and transport processes, and risk. Those differences will play an important role, and will pose unique challenges, when developing and assessing remedial alternatives in the FS. For example, conventional parameters and contaminants that act as non-COPEC stressors, including low DO, elevated porewater sulfide, and bulk sediment concentrations of complex hydrocarbon mixtures (i.e., C19-C36), may contribute to the increased toxicity observed in sediment toxicity tests in some tributary areas where these stressors are elevated, but where porewater concentrations of COPECs are below risk thresholds. Figure 8-3 presents key components of the CSM in the context of these three broad reaches, described in the rest of Section 8.

In the rest of this section, the CSM is constructed using data collected as part of the RI dataset, as well as other sources of information, including hydrodynamics, sediment transport, chemical fate and transport, and bioaccumulation, and considers the geographic, temporal, and spatial variability of Newtown Creek. The CSM and associated risk assessments will ultimately form the basis for development and selection of a remedial action.

8.2 Site Setting

The land use around Newtown Creek from the 1800s through the present has been predominately industrial. This industrial development occurred in parallel with municipal use of Newtown Creek as a receiving waterbody of both stormwater and wastewater discharges. A perspective on the various industrial activities along Newtown Creek is provided in Section 3.2.6. The municipal use of Newtown Creek as a receiving waterbody for stormwater and sanitary and industrial wastewater discharges has evolved over time, especially with the initial construction of a WWTP in the late 1960s. Newtown Creek continues to be a major receiving waterbody of industrial and municipal stormwater discharges and CSO discharges (containing flows of stormwater, as well as sanitary and industrial wastewater) as well as treated effluent from the Newtown Creek WWTP overflow during rainfall events. Newtown Creek is also a designated SMIA, which will continue to give preference to industrial uses in upland areas. Consequently, the land use history and urban landscape in which Newtown Creek exists shapes the CSM and informs the nature and extent of COPCs and potentially significant sources, as well as key fate and transport characteristics, pathways, and exposure scenarios. Notwithstanding its SMIA designation, changes in land use patterns may occur over time in upland areas adjacent to the Study Area, and these changes may result in changes to some inputs to the creek. It is clear that industrial and commercial land uses will continue to dominate the upland areas around Newtown Creek well into the future.

Newtown Creek is a catchment waterbody, surrounded by an urban landscape and open only to the East River. A number of tributaries, or subcatchments, feed the main channel of Newtown Creek. The East River has a dominant effect through twice-daily tidal exchanges that move suspended solids in and out of Newtown Creek, resulting in deposition, resuspension, and mixing of particles in surface sediment. In addition, urban runoff and point source discharges throughout Newtown Creek, especially the MS4s and CSOs in the tributary subcatchment areas, directly introduce suspended solids that are predisposed to settle onto the surface sediment bed near where these discharges occur.

The modifications to Newtown Creek that have occurred over time have resulted in a system that is largely engineered for industrial, municipal, and navigational purposes. Mudflat habitat available for wildlife use ranges from 0% of the Study Area at high tide to approximately 5% of the Study Area at low tide and is mostly accounted for by the sediment accumulations near major point source discharges at the headwaters of Maspeth Creek (i.e., CSO sediment mounds). The subtidal surface sediment generally is a low-quality habitat for benthic macroinvertebrates, further limited by periodic low DO conditions in the summer in the tributary subcatchment locations.

This RI uses samples collected from all 14 Phase 1 reference areas, which are tidal inlets in the New York Harbor area. These data reflect four categories of urban condition: Industrial/CSO, Industrial/Non-CSO, Non-Industrial/CSO, and Non-Industrial/Non-CSO. Surface sediment data are available for the 14 Phase 1 reference areas (indicated by boxes in Figure 8-4). Surface water and tissue data, as well as bioassay results and benthic community surveys, are also available for the four Phase 2 reference areas (one for each of these urban condition categories; indicated by blue boxes in Figure 8-4).

In addition, the Study Area will continue to be subject to ongoing influences from human activities that are specific to: 1) this urban environment, in the form of ongoing discharges of solids from CSO, MS4, and other point source and stormwater discharges, particularly in the tributaries; and 2) its industrial nature, which includes navigation activities, historical and current discharges from upland properties, and shoreline seeps including NAPL seeps as well as other ongoing sources of contaminants (see Figure 8-1). To the extent that ongoing sources are not fully controlled, they will continue to influence the future of the system.

8.3 Physical Characteristics of the Study Area

Newtown Creek is a tidal inlet to the East River, with no current natural tributary inflows. The main channel of the creek extends inland approximately 3.8 miles, with five side channels, or tributaries, entering at various points along its length (see Figure 8-5). A navigation channel runs through the main stem and extends into the tributaries. All five tributaries tend to be narrower than the main channel and have shallower depths.

The hydrodynamics of the Study Area are dominated by twice-daily tides and by rainfall-related flows from point sources and overland flow. Tidal mixing with East River water is most pronounced in CM 0 – 2 of the main stem, but continues beyond CM 2 (see Section 6.2.2; see also Section 4 of Appendix G). Under dry weather conditions, flows to the Study Area include point source discharges of treated effluent from groundwater remediation and dewatering systems; less frequent municipal discharges (e.g., bypasses that may occur when combined sewer infrastructure malfunctions [NYCDEP 2011c, 2012b, 2013a, 2014b, 2015, 2016a]); and in some areas, from groundwater discharge into subsurface sediment from the underlying native material and groundwater discharge to surface water through lateral flow through vertical permeable shorelines (defined as pile-supported concrete, precast concrete blocks, and vertical wood bulkheads). Hydrodynamic processes (i.e., tidal currents and density-driven circulation) generate relatively low, near-bed current velocities throughout large portions of the Study Area, which result in minimal or near-zero erosion of the sediment bed. The upper layer of the sediment bed is reworked and resuspended due to biological and physical processes, including localized resuspension due to vessel propwash in the navigation channel of Newtown Creek, the Turning Basin, Whale Creek, and the lower portion of English Kills (see Section 6.3; see also Section 5.3 of Appendix G). Additionally, erosion and subsequent deposition of sediments occur at smaller spatial scales in the area of larger CSO discharges.

During wet weather, the Study Area receives water from overland flow and point source discharges, especially at the heads of the relatively shallower tributaries. These discharges include large CSOs, the WWTP treated effluent overflow outfall in Whale Creek, and industrial and municipal stormwater and overland flow during storms. These are the significant sources of inflow during wet weather, and they contain not only CERCLA hazardous substances, but also contribute non-COPEC stressors. During certain wet weather events, freshwater inflows in the tributaries can cause resuspension of the upper layer of the sediment bed in localized areas, as well as salinity stratification of the water column that typically lasts less than 24 hours, before the freshwater mixes with the estuarine waters of the creek.

Suspended solids are introduced into the Study Area water column primarily by inflows from the East River, CSOs, stormwater, and the Whale Creek WWTP treated effluent overflow (see Figure 5-13 and Section 5.4.3 of Appendix G). These solids are transported and mixed within the surface water and deposited on the sediment bed. The depositing solids continuously add to the existing sediment bed, burying and isolating the majority of the contaminants historically deposited in the sediment bed. The sediment bed throughout Newtown Creek is a cohesive (muddy) bed that is primarily net depositional, due to the low near-bed current velocities. Several LOEs demonstrate the net depositional nature of the Study Area, with net deposition rates ranging from less than 1 cm/year to as much as 7 cm/year depending on the location and LOE used (see Figures 8-6 and 8-7 and Section 5.2.1 of Appendix G). Dredging in the navigation channel, if performed in the future, has the potential to expose sediment containing higher contaminant concentrations buried at depth, depending on the depth of dredging and how the overall project is implemented (e.g., placement of cover material can be used to prevent long-term exposure of the deeper sediments following dredging). Propwash scour due to vessel traffic can cause localized resuspension of sediment, but modeling evaluations completed to-date indicate that in most cases the resuspended sediments deposit back to the bed within the same general area where they originated.

Depositing solids mix with the existing near-surface sediment. This mixing is influenced by a number of processes, including biological activity in the surface sediment, propwash due to marine vessel traffic in the navigation channel, and high velocity point source discharges in localized areas during wet weather events. This mixing process, coupled with ongoing sedimentation, has resulted in a continual blending of newly deposited sediment with the surface sediment, such that historically elevated COPC concentrations associated with legacy contamination are still reflected in the sediment surface today in several reaches of the Study Area. This mixing process varies spatially within the Study Area (i.e., by reach) and has the greatest effect on surface sediment concentrations in areas where NSRs are relatively lower and subsurface chemical concentrations are substantially higher than those in surface sediment, such as CM 2+.

In summary, based on these physical characteristics, Newtown Creek is evaluated in this RI in the following three reaches (see Figure 8-5):

- The lower main stem, from the mouth to approximately CM 2 (CM 0 – 2), is dominated by extensive tidal exchange with the East River. Depositing solids originate primarily from the East River, with contributions from ongoing point source discharges increasing with distance upstream from the East River. Figure 8-8 shows the proportions of point source and East River solids depositing in each reach of the Study Area, based on the results of the sediment transport model. As shown in Figures 8-6 through 8-8, NSRs vary from approximately 1 to 5 cm/year, with minor changes in NSRs during the last 50 to 75 years. This reach has an average width of approximately 100 meters and an average depth at mean tide level of 5.6 meters.

- The upper main stem, including the Turning Basin (CM 2+), is a more complex portion of the Study Area. Depositing solids originate both from downstream (the East River) and upstream (primarily CSOs and stormwater outfalls). NSRs estimated over a 10- to 25-year horizon (see Figure 8-6) vary from approximately 0.5 to 1 cm/year. NSRs estimated over the last 50 to 75 years are greater, approximately 2.5 cm/year, due to changes in CSO sediment loads over time (see Section 5.2.1 of Appendix G). This reach has an average width of approximately 120 meters and an average depth at mean tide level of 4.9 meters.
- The tributaries exhibit low surface water current velocities under typical conditions. Storm-related point source inflows provide nearly all the solids that deposit on the sediment bed in the upper tributaries (i.e., Maspeth Creek, East Branch, and English Kills). Although the tributaries are considered as one broad area in certain portions of the discussion in this section, it should be noted that each tributary differs in circulation, deposition characteristics, and solids sources. NSRs estimated over a 10- to 25-year horizon (see Figure 8-6) vary from approximately 0.5 to 4 cm/year in the tributaries. NSRs estimated over the last 50 to 75 years are greater, varying from approximately 2.5 to 7 cm/year, due to changes in CSO sediment loads and sediment trapping efficiency over time. Each tributary differs in geometry; the approximate average widths and depths at mean tide level are as follows:
  - Dutch Kills: width of 53 meters; depth of 2.8 meters
  - Whale Creek: width of 46 meters; depth of 5.4 meters
  - Maspeth Creek: width of 66 meters; depth of 0.81 meter
  - East Branch: width of 58 meters; depth of 3.3 meters
  - English Kills: width of 51 meters; depth of 3.6 meters

8.4 Nature and Extent of Contamination

The discussion on the nature and extent of contamination in Section 4 evaluated each sampling medium for the following eight contaminants or groups of contaminants (depending on the availability of data in the medium): TPAH (17), TPAH (34), C19-C36, TPCB, 2,3,7,8-TCDD, dieldrin (only for certain media based on risk assessment findings), Cu, and Pb. As discussed in the comparative evaluation presented in Section 4.12,[290] chemicals from the same general class—i.e., hydrocarbons (TPAH [17], TPAH [34], and C19-C36), bioaccumulative organics (TPCB, 2,3,7,8-TCDD, and dieldrin), and metals (Cu and Pb)—were found to correlate with respect to spatial patterns, with some differences on smaller and localized scales. Although the chemicals within a given class generally correlate spatially, this does not mean they contribute equally to risk. Based on the results of the risk assessments and the Section 4.12 comparative evaluation, the RI Report evaluations of sources and fate and transport focus on TPAH (17), TPCB, and Cu as representative contaminants from each of the three classes (i.e., hydrocarbons,[291] bioaccumulative organics, and metals). Therefore, the following discussion with respect to the CSM also focuses on these same three contaminants.[292] As described in Section 4, TPAH (34), C19-C36, 2,3,7,8-TCDD, Pb, and dieldrin also contribute to risk (to varying degrees) but were not included in Sections 5 and 6 and therefore are not the focus of the CSM. However, because 2,3,7,8-TCDD was identified as a primary risk driver for human health from the consumption of blue crab, it is also included in the following discussion for surface sediment. In addition,

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[290] Comparisons between TPAH (17) and TPAH (34) are presented in Section 4.2.5.1.

[291] Due to the observed differences in the comparative evaluations for hydrocarbons, TPAH (17) cannot be considered representative of hydrocarbons for the purposes of the discussions of nature and extent of contamination. Likewise, although it was considered representative of hydrocarbons for the RI evaluations of sources and fate and transport, TPAH (17) should not be used as a surrogate for other hydrocarbons in future evaluations, so the FS will consider the other hydrocarbon groups (i.e., TPAH [34] and C19-C36) individually.

[292] As described in Sections 4.1.2 and 4.12, the RI Report focuses on TPAH (17), TPCB, and Cu for a number of reasons. TPAH (17) is a primary risk driver in the BERA (see Appendix I), TPCB is a primary risk driver in the BHHRA and BERA (see Appendices H and I, respectively), and Cu was selected as a representative metal because the BERA estimated HQs greater than 1 (but less than 3) for some receptors. TPAH (34), C19-C36, 2,3,7,8-TCDD, and Pb are included in the nature and extent evaluation for all media (when analyzed) in Section 4, because TPAH (34) and C19-C36 are important contributors to benthic toxicity (as determined by USEPA subsequent to completion of the BERA), 2,3,7,8-TCDD was identified as a risk driver in the BHHRA and as a COPEC in the BERA, and Pb was identified as a COPEC in the BERA. Although dieldrin was not identified as a COPC or COPEC, it was included in the nature and extent evaluation of surface sediment (Section 4.2) and tissue (Section 4.10), because of elevated concentrations in polychaete tissue in one reach of the Study Area (English Kills). However, these five contaminants were not included for evaluations of sources and fate and transport in Sections 5 and 6, respectively, because: 1) the distributions in environmental media (including surface sediment) are broadly similar to those of within the same class (i.e., hydrocarbons, bioaccumulative organics, and metals); 2) in some of the locations or some of the media, some of these contaminants (e.g., C19-C36, 2,3,7,8-TCDD, and Pb) were either not analyzed or were infrequently detected (in the case of surface water, porewater, and groundwater—these contaminants were detected at generally high frequencies in sediment); and 3) their fate and transport characteristics (i.e., partitioning behavior) are similar, especially to others in the same class, such that they can be represented by TPAH (17), TPCB, and Cu.

the nature and extent of NAPL is included in Section 8.4.1 for surface and subsurface sediment and in Section 8.4.2 for native material.

8.4.1 Surface and Subsurface Sediment

8.4.1.1 TPAH (17), TPCB, and Cu

Patterns in the distribution of contaminants in the surface sediment (defined operationally as a depth of 0 to 15 cm [0 to 6 inches]) and subsurface sediment (from 15 cm [6 inches] depth to the interface with the underlying native material) of the Study Area have been observed in the RI data (see Sections 4.2 and 4.3). Overall, the spatial patterns for TPAH (17), TPCB, and Cu generally show increasing surface and subsurface contaminant concentrations when moving from downstream reaches into the upstream reaches and tributaries. In addition, the concentrations of TPAH (17), TPCB, and Cu in subsurface sediment are higher than in surface sediment (see Section 4.3, Figures 4-64, 4-76, and 4-82). This subsection summarizes the data by reach and shows box plots for TPAH (17), TPCB, and Cu in surface sediment in Figures 8-9, 8-10, and 8-11. Figures for subsurface sediment are not presented, because the spatial patterns are similar to those for surface sediment (see Section 4.3). This subsection also discusses the concentrations of these contaminants in shoreline samples collected in the FS Part 1 field program (more detailed discussion can be found in Section 5.4.2).

Surface sediment concentrations in all reaches are compared to surface sediment data collected in reference areas (including data collected during Phase 1 at the 14 Phase 1 reference areas and data collected during Phase 2 at the 4 Phase 2 reference areas). Notable findings are provided on a reach-specific basis:

- **CM 0 – 2**
  - Concentrations of TPAH (17), TPCB, and Cu in surface sediment are lower than those of surface sediments in upstream reaches, and concentrations measured in CM 0 – 1 are lower than those in CM 1 – 2 (see Figures 8-9, 8-10, and 8-11).
  - Concentrations of TPAH (17) in the subsurface sediment in CM 1 – 2, including Whale Creek, are higher than in CM 0 – 1 (but lower than CM 2+) and more consistent with concentrations observed in the subsurface sediment of the upstream tributaries (see Section 4.3, Figure 4-64).

  - Concentrations of TPCB in the subsurface sediment in CM 1 – 2 are higher than in CM 0 – 1, but lower than subsurface sediment concentrations in CM 2+ and English Kills (see Section 4.3, Figure 4-76).
  - Concentrations of Cu in the subsurface sediment in CM 1 – 2 are higher than in CM 0 – 1, but lower than subsurface sediment concentrations in CM 2+ and the tributaries (see Section 4.3, Figure 4-84).
  - Concentrations of TPAH (17), TPCB, and Cu measured in surface sediment from CM 0 – 1 are consistent with those in reference areas with similar characteristics related to the degree of industrialization and CSO inputs (see Figures 8-9, 8-10, and 8-11).
  - Concentrations of TPAH (17), TPCB, and Cu in surface sediment from CM 1 – 2 show considerable overlap with reference area data for those reference areas that are in the Industrial/CSO category, although the majority of the samples from CM 1.5 – 2 have TPCB and Cu concentrations that exceed the upper end of the range of the reference area data. TPAH (17) concentrations only exceed the upper end of the reference area data in a few locations.
  - Concentrations of TPAH (17), TPCB, and Cu in shoreline sediment samples are generally consistent with the range of concentrations observed in other surface sediment data in the area from where the shoreline samples were collected. The one exception occurs for the sample collected from the shoreline adjacent to LIRR – 47th Avenue; TPAH (17) concentrations are lower than the other surface sediment data in CM 1 – 2 for this sample.
  - Concentrations of contaminants, including TPAH (17), TPCB, and Cu, tend to increase with depth in the sediment, indicating that surface sediment concentrations have been declining over time, in part due to deposition of less contaminated solids (see Section 4.3, Figures 4-66, 4-78, and 4-86).

- **CM 2+**
  - Overall, the highest observed concentrations of TPAH (17), TPCB, and Cu in surface and subsurface sediment occur in CM 2+ (see Figures 8-9, 8-10, and 8-11 for surface sediment, and Section 4, Figures 4-64, 4-76, and 4-84 for subsurface sediment, respectively).

  - Concentrations of TPCB in surface and subsurface sediment are higher in both CM 2+ and English Kills than in the remainder of the Study Area (see Figures 4-76 and 8-10).
  - Concentrations of TPAH (17), TPCB, and Cu in surface sediment generally exceed the reference area concentrations, indicating the presence of site-specific historical and ongoing contaminant sources (see Figures 8-9, 8-10, and 8-11).
  - Concentrations of TPAH (17), TPCB, and Cu in shoreline samples are generally consistent with or lower than the range of concentrations observed in other surface sediment data, with a limited number of shoreline samples that contain higher concentrations of both TPAH (17) and TPCB. For example, one of the two samples collected from shoreline sediment adjacent to Cipico Construction Inc. contains the highest TPAH (17) concentrations measured in surface and subsurface sediment. In addition, higher TPCB concentrations were measured in shoreline subsurface samples adjacent to LIRR 47th Avenue and Cipico Construction Inc., while higher Cu concentrations were also measured in shoreline subsurface samples adjacent to Cipico Construction Inc.
  - Concentrations of contaminants, including TPAH (17), TPCB, and Cu, tend to increase with depth in the sediment (see Figures 4-66, 4-78, and 4-86, respectively).

- **Tributaries**
  - Concentrations of TPAH (17), TPCB, and Cu in surface sediment are generally higher than in CM 0 – 2, although the tributaries differ in their longitudinal patterns of contaminant concentrations. For example, concentrations of the chemicals in the lower 0.5 mile of English Kills are among the highest of the tributaries. TPCB concentrations are also elevated in Dutch Kills, where the highest reported surface sediment concentration (380 mg/kg) in the entire Study Area was measured.
  - Concentrations of TPAH (17) in subsurface tributary sediment are lower than those in CM 2+, similar to those in CM 1 – 2, and higher than those in CM 0 – 1 (see Figure 4-64).
  - Concentrations of TPCB in subsurface sediment in English Kills are similar to those in CM 2+ and higher than those in Maspeth Creek, East Branch, and CM 0 – 2 (see Figure 4-76).

- Concentrations of Cu in subsurface sediment from Maspeth Creek are similar to those in CM 2+, but are higher than in subsurface sediments from the other tributaries and CM 0 – 2 (see Figure 4-84).
- Concentrations of TPAH (17), TPCB, and Cu in surface sediment generally exceed the reference area concentrations, as a result of the mixing of ongoing sources with residual historical contamination (see Figures 8-9, 8-10, and 8-11).
- Surface sediment exhibits higher TOC levels than are found in CM 0 – 2 and in the East River main stem reference areas, likely primarily due to discharges of solids from CSO and MS4 point sources (see Figure 4-9).
- Concentrations of TPAH (17), TPCB, and Cu in tributary shoreline samples are generally within the range or lower than the concentrations observed in other surface sediment data, with a few exceptions. For example, relatively high concentrations of TPAH (17), TPCB, and Cu occur in samples collected from the shorelines within English Kills adjacent to LIRR – Johnson Avenue; within Dutch Kills adjacent to Borden Realty, Co.; and within the sediment mound outside the NCB-015 CSO outfall (EK135) and adjacent to Frito Lay II (EK133).
- Concentrations of contaminants, including TPAH (17), TPCB, and Cu, tend to increase with depth in sediment (see Figure 4-66, 4-78, and 4-86, respectively).

8.4.1.2 2,3,7,8-TCDD

2,3,7,8-TCDD concentrations in surface sediment of the Study Area and the four Phase 2 reference areas are shown in Figure 8-12. Overall, the spatial pattern of 2,3,7,8-TCDD is similar to that of TPCB (see Figure 8-10), with the exception of Dutch Kills and Whale Creek. For both 2,3,7,8-TCDD and TPCB, surface sediment concentrations in CM 0 – 2 are lower than those in CM 2+ and English Kills. Similarly, surface sediment concentrations in Maspeth Creek and East Branch are less than those in CM 2+ and English Kills. However, 2,3,7,8-TCDD surface sediment concentrations in Whale Creek are higher than those in Dutch Kills, whereas for TPCB, the reverse is true. While the surface sediment concentrations of both contaminants Study Area-wide are higher than those of the reference areas, this difference is more pronounced for TPCB than 2,3,7,8-TCDD.

8.4.1.3 NAPL

The presence and extent of NAPL in sediment were extensively investigated during the RI and FS Part 1 field programs, which included multiple field investigations and the collection of hundreds of surface sediment grabs and cores used to evaluate NAPL in Study Area surface and subsurface sediment, and native material. The presence of NAPL blebs or a NAPL layer in a shake test indicates that NAPL is present. The lack of NAPL blebs or a NAPL layer (i.e., no observation, or sheen only) confirms that NAPL is not present.

For much of the Study Area, where NAPL was observed in sediment, it was primarily in a residual state (i.e., shake test bleb results, visual observations of NAPL blebs in sediment) and was distributed intermittently (i.e., located sporadically throughout an area and not clustered at a particular location).[293] A relatively greater magnitude of NAPL (i.e., shake test layer results, along with visual observations of NAPL coated and/or NAPL saturated sediment), referred to as Category 2/3 NAPL, was observed in three discrete areas of the Study Area, referred to as Category 2/3 Areas.

Based on the RI methods for defining NAPL (Anchor QEA 2014b), notable findings regarding the data are as follows:

- **CM 0 – 2**
  - NAPL was not observed in surface sediment; however, sheen was observed in surface sediment during sample processing at a limited number of scattered locations (27 of 111 locations).
  - In subsurface sediment, sheen and NAPL were observed at scattered, discontinuous locations. Where NAPL was observed (34 of 66 locations), it was generally intermittent and in a residual state (31 of 34 locations).

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[293] Residual NAPL is the condition defined as NAPL saturation that is sufficiently low so that the NAPL consists of discrete blebs, trapped by capillary forces, and is, therefore, immobile. This classification is specific to the ability of the NAPL to advect (i.e., flow) as a nonaqueous fluid phase. The interpretation that blebs represent residual, immobile NAPL is based on the observation that in core samples, the blebs are present as small, discrete droplets, but for the most part, the sediment is not visually contaminated. This matches the description of residual NAPL as described in the literature (Schwille 1988; Cohen and Mercer 1993; Pankow and Cherry 1996; API 2003; ITRC 2004; Sale et al. 2008; ITRC 2009; Kueper and Davies 2009). The NAPL mobility testing program performed in the FS Part 1 field program has demonstrated that NAPL in CM 0 – 2 is immobile.

  - From CM 1.6 to 1.7, a limited number of cores produced shake test layer results, indicating the presence of Category 2/3 NAPL (3 of 16 locations in CM 1.6 to 1.7). This area is referred to as the CM 1.7 Category 2/3 Area.
  - NAPL mobility evaluation (using centrifuge testing) of 24 CM 0 – 2 subsurface sediment samples, including samples collected from the CM 1.7 Category 2/3 Area, demonstrated that where present, NAPL is immobile (see Sections 5.3.4 and 5.4.1 of Appendix C).

- **CM 2+**
  - NAPL was not reported in the majority of surface sediment collected from this reach (122 of 132 locations), although sheen was observed in surface sediment during sample processing at a number of locations (72 of 132 locations), particularly at the upstream portion of the reach.
  - Where NAPL was observed in surface sediment, it was generally intermittent and in a residual state (7 of 10 locations) and located upstream of CM 2.4, within the area referred to as the Turning Basin Category 2/3 Area.
  - NAPL and sheen were observed in subsurface sediment in a number of cores collected in this reach (50 of 89 locations and 30 of 89 locations, respectively). Residual NAPL and/or sheen were observed in the majority of cores from CM 2 to 2.4. Upstream of CM 2.4 in the Turning Basin Category 2/3 Area, a combination of residual and Category 2/3 NAPL was observed in subsurface sediment, at varying depths and thicknesses.
  - Quantitative NAPL mobility testing for CM 2+, including the Turning Basin Category 2/3 Area, is being performed as part of the FS Part 2 field program (data for the FS Part 2 field program are not included in the RI Report and are presented in the FS NAPL DER [Anchor QEA 2022a]).

- **Tributaries**
  - NAPL was observed in surface sediment in only 1 of 256 locations sampled in the tributaries (lower English Kills).
  - Sheen was observed in approximately half of the surface sediment samples (130 of 256 locations), with observations scattered throughout the tributaries.

- NAPL was not observed in subsurface sediment in Dutch Kills or Whale Creek; however, sheen was observed in subsurface sediment in more than half of the cores collected from these areas (20 of 30 locations).
- NAPL was observed in subsurface sediment in a limited number of locations scattered in Maspeth Creek, East Branch, and the upper reach of English Kills (11 of 41 locations), while sheen was observed in subsurface sediment at a number of locations in these areas (22 of 41 locations). NAPL was more widely observed in subsurface sediment in the lower reach of English Kills than in the other tributaries (16 of 43 locations), and where NAPL was observed, it was primarily in a residual state (11 of 16 locations). In the lower reach of English Kills, sheen was observed in approximately half of the subsurface sediment cores (21 of 43 locations).
- Category 2/3 NAPL was observed in subsurface sediment in a limited number of cores (5 of 27 locations), all located in the lower portion of English Kills between CM 2.95 and 3.2. This area is referred to as the Lower English Kills Category 2/3 Area.
- Quantitative NAPL mobility testing for the tributaries, including the Lower English Kills Category 2/3 Area, is being performed as part of the FS Part 2 field program (data for the FS Part 2 field program are not included in the RI Report and are presented in the FS NAPL DER [Anchor QEA 2022a]).

8.4.2 Native Material

8.4.2.1 TPAH (17), TPCB, and Cu

Native material differs substantially from subsurface sediment. OC content (as quantified by TOC, TPH, and soot carbon concentrations) is low compared with surface and subsurface sediment, except at a limited number of locations. TPAH (17) concentrations in native material are generally two to three orders of magnitude lower than those in subsurface sediment, except at a limited number of locations. Of the TPCB samples in native material, 66% were non-detect, and detected TPCB concentrations are generally two to three orders of magnitude lower than in subsurface sediment. Cu concentrations in native material are generally one to two orders of magnitude lower than in subsurface sediment and do not exhibit notable patterns. The median concentration of Cu in native material (22 mg/kg) is similar to geological background concentrations documented in the literature (21 mg/kg; see Section 4.4.3.6).

Notable findings regarding the data are as follows:

- **CM 0 – 2**
  - A limited number of scattered samples exceed 10 mg/kg TPAH (17) (a level that is less than the median surface sediment concentrations from several of the reference areas; see Figure 8-9).
- **CM 2+**
  - A number of samples exceeding 10 mg/kg TPAH (17) are found in CM 2+, where higher subsurface sediment concentrations are also found. These native material concentrations are within the range of subsurface sediment concentrations and lie within the same footprint.
  - A limited number of samples exceeding 1 mg/kg TPCB are found in CM 2+, between CM 2.5 and 2.7, where higher subsurface sediment concentrations are also found. These native material concentrations are within the range of subsurface sediment concentrations and lie within the same footprint.
  - Elevated Cu in the Turning Basin was observed in one sample (14,000 mg/kg).
- **Tributaries**
  - A limited number of scattered TPAH (17) samples exceed 10 mg/kg. In lower English Kills, elevated TPAH (17) concentrations are found in native material where higher subsurface sediment concentrations are also found. These native material concentrations are within, or somewhat above, the range of subsurface sediment concentrations and lie within the same footprint.
  - In Dutch Kills there is one sample that shows elevated concentrations in the native material for both TPCB (6.9 mg/kg) and Cu (1,600 mg/kg).

8.4.2.2 NAPL

NAPL observations in the native material were primarily limited to the areas of the Turning Basin and English Kills, where NAPL was also observed in subsurface sediment. Isolated sheens were infrequently observed in the native material in the main stem (primarily between CM 1.3 and 2.7), in the lower English Kills, and in one location in Maspeth Creek. Notable findings regarding the data are as follows:

- **CM 0 – 2**
  - Residual NAPL was observed in native material near CM 1.7; this was the only NAPL observed in native material in this reach (1 of 43 locations).
  - Sheens were observed in native material at isolated locations throughout the reach (7 of 43 locations). Sheen observations generally exhibited no clear spatial pattern, except between CM 1.6 and 1.7, where sheen was observed more frequently in native material (3 of 13 locations).
  - NAPL mobility evaluation (using centrifuge testing) was performed on four native material samples, including one sample near CM 1.7. Where present, NAPL was immobile.
- **CM 2+**
  - NAPL was observed in native material at 24 of 73 locations in this reach, while sheen was observed in native material at 7 of 73 locations. Sheen and primarily residual NAPL were observed from CM 2 to 2.5 (4 of 32 locations and 4 of 6 locations, respectively). Upstream of CM 2.5 in the Turning Basin Category 2/3 Area, the majority of NAPL observed was in the form of Category 2/3 NAPL (15 of 18 locations).
  - Quantitative NAPL mobility testing for CM 2+, including the Turning Basin Category 2/3 Area, is being performed as part of the FS Part 2 field program (data for the FS Part 2 field program are not included in the RI Report and are presented in the FS NAPL DER [Anchor QEA 2022a]).
- **Tributaries**
  - Sheen and NAPL were observed to a limited degree in native material (4 of 84 locations and 18 and 84 locations, respectively).
  - All the NAPL observed in native material was located in the lower portion of English Kills between CM 2.95 and 3.2 (Lower English Kills Category 2/3 Area). Between CM 2.95 and 3.2, the majority of NAPL observed was in the form of Category 2/3 NAPL (14 of 18 locations).

  - Quantitative NAPL mobility testing for the tributaries, including the Lower English Kills Category 2/3 Area, is being performed as part of the FS Part 2 field program (data for the FS Part 2 field program are not included in the RI Report and are presented in the FS NAPL DER [Anchor QEA 2022a]).

8.4.3 Surface Water

Surface water is potentially a significant exposure source for some species in the aquatic food web of Newtown Creek. Dry weather conditions are the most common conditions occurring throughout the year and characterize most of the exposure of the water column-associated food web. Contaminant sources to the surface water under dry weather conditions include East River tidal water, flux from the surface sediment, treated effluent from groundwater remediation and dewatering systems, and less frequent municipal discharges (e.g., bypasses that may occur when combined sewer infrastructure malfunctions [NYCDEP 2011c, 2012b, 2013a, 2014b, 2015, 2016a]). Contaminant concentrations of TPAH (17), TPCB, and Cu in surface water under dry weather conditions exhibit little to no variation with season or tide conditions, with some spatial variation (increasing concentrations downstream to upstream) for TPAH (17) and TPCB, but little to no spatial variation for Cu. This limited spatial pattern is primarily due to mixing from tidal currents and to the influence of the East River. Higher concentrations of some contaminants are observed in the surface water of some of the upper tributaries. Flux to surface water from surface sediment porewater (due to advection, diffusion, and tidal exchange) is likely a contributing factor to elevated concentrations in these tributaries. Other sources, such as lateral groundwater discharge, shoreline seeps, bank erosion, and gas ebullition (see Figure 8-1) may also affect surface water concentrations on localized spatial and temporal scales.

During wet weather, discharges from CSO, MS4, and other point sources produce measurable loads of contaminants to the Study Area, in addition to the sources present during dry weather described in the preceding paragraph. These discharges result in increased concentrations of TPAH (17), TPCB, and Cu in surface water throughout the Study Area, relative to dry weather. These discharges provide short-lived exposure to the aquatic food web, but at higher concentrations than under dry weather conditions, and also lead to the deposition of contaminant solids onto the sediment bed.

Notable findings regarding the surface water data include the following (see Figure 8-13):

- **CM 0 – 2**
  - Dry weather TPAH (17) concentrations are generally within the range of the East River data and within the range of the data from the four Phase 2 reference areas. These comparisons suggest that there is limited Study Area-specific influence in this reach under dry weather conditions (i.e., that TPAH [17] sources that are common to the urban environment exert a dominant effect on the surface waters of Newtown Creek in this reach). TPAH (17) concentrations during wet weather are 2 to 3 times higher than those for dry weather in this reach.
  - Dry weather TPCB concentrations throughout CM 0 – 2 are generally within (or slightly above) the range of the East River data (medians for CM 0 – 1 and CM 1 – 2 are within approximately 50% of the median for East River data). Both CM 0 – 2 and East River TPCB concentrations exceed those measured in the four Phase 2 reference areas. Wet weather TPCB concentrations are 1.5 to 2 times higher than those for dry weather in this reach.
  - In contrast, dry weather Cu concentrations are almost entirely within range of both the East River and Phase 2 reference area data, suggesting that general urban Cu sources also control Cu concentrations in the surface waters. Differences between wet and dry weather concentrations are less than a factor of 2 for Cu in this reach.
- **CM 2+**
  - Dry and wet weather surface water concentrations in CM 2+ are generally similar to those in CM 0 – 2. Flux from surface sediment porewater and resuspension, along with contribution from wet weather point source discharges, likely contributes to surface water concentrations in this reach. The contributions from each source will be quantified as part of the chemical fate and transport modeling.
- **Tributaries**
  - Compared to the main stem, higher concentrations are observed in the surface waters in the tributaries in many cases, under both dry and wet weather conditions. For example, TPCB concentrations in English Kills (where elevated TPCB concentrations in surface sediment were observed, primarily in the lower 0.5 mile of the tributary) are generally higher than the rest of the Study Area.

> This, coupled with other fate and transport evaluations (see Section 8.6), suggests that exchange flux from surface sediment porewater contributes to surface water contamination in these upstream reaches. Similarly, the distributions of wet weather TPAH (17) concentrations are generally higher in Maspeth Creek, East Branch, and English Kills, as compared to the main stem, reflecting effects from point source discharges (based on presence of large point source discharges in these tributaries and sampling of point sources; see Section 5 of Appendix E). The contributions from sources will be quantified as part of the CFT modeling being conducted for the FS. There is also variation among the tributaries and chemicals. For example, arithmetic average TPAH (17) concentrations under wet weather are higher than those under dry weather by factors of 4 to 5 in Dutch Kills and Maspeth Creek and factors of 2 to 3 in English Kills and East Branch. Differences between arithmetic average TPCB and Cu concentrations are generally a factor of 2 or less between wet and dry weather in all four tributaries sampled (note: Whale Creek was not sampled during wet weather).

Using partitioning theory (see Section 4.1.3.5), the surface water chemical concentration data were combined with measurements of TSS to calculate particulate phase concentrations that are directly comparable to sediment data. The spatial patterns in particulate phase TPAH (17), TPCB, and Cu concentrations show similar patterns to those observed in whole-water samples:

- The lowest concentrations tend to be in CM 0 – 1 (i.e., the CM nearest the East River).
- Concentrations tend to increase with increasing distance upstream in the main stem of Newtown Creek.
- Concentrations generally tend to be higher in the more upstream tributaries.

Particulate phase concentrations in Study Area surface water also show increases by factors of 2 to 4 during wet weather, as compared to dry weather.

Surface water particulate phase concentrations in the Study Area are typically higher than those observed in the reference area surface water data, but are generally lower than those in Study Area surface sediment. With ongoing sedimentation, this is expected to reduce surface sediment concentrations over time.

8.4.4 Groundwater and Porewater

The magnitude and direction of groundwater flow into the Study Area was quantified through deployment of seepage meters by USGS and measurement of vertical hydraulic gradient using piezometers and monitoring wells. Groundwater samples from 64 temporary monitoring wells installed in native material characterized the spatial distribution of dissolved chemicals beneath the Study Area. The vertical distribution of concentrations was characterized primarily based on these data and shallow (0- to 15-cm [0- to 6-inch] and 15- to 30-cm [6- to 12-inch] depth intervals) porewater samples. Notable findings regarding these data are summarized as follows:

- **CM 0 – 2**
  - Zones of negative seepage—where surface water flows from the Study Area into groundwater—were identified near CM 0.5, CM 1.1, and between CM 1.5 and 2.0. This interpretation is supported by seepage data, locations of known groundwater extractions, salinity and chloride concentrations, and specific conductance measured in groundwater beneath and adjacent to the creek in these areas (see Sections 3.7.1 and 5.3.3.3.1 of Appendix F). Between these reaches, groundwater discharges to the base of the Study Area (positive seepage) and directly to surface water.
  - TPAH (17) concentrations in CM 0 – 2 are generally higher in groundwater than in shallow porewater. TPAH (17) concentrations in groundwater are variable in this reach, with higher concentrations in CM 0 – 1 and lower in CM 1 – 2.
  - TPCB concentrations in groundwater beneath CM 0 – 2 are relatively low (i.e., less than 10 ng/L in all but three samples), and generally similar to, or lower than, those in shallow porewater.
  - Cu concentrations are also variable in this reach. In CM 0 – 1, data are non-detect in all groundwater samples, and Cu concentrations are low in shallow porewater. Although Cu concentrations are relatively low in groundwater throughout the Study Area, CM 1 – 2 had the highest arithmetic average Cu concentration of any reach, and the arithmetic average Cu concentration in groundwater is higher than that in shallow porewater by a factor of 4.

- **CM 2+**
  - Groundwater discharges to the base of the Study Area (from the underlying native material) in this reach. This conclusion is supported by positive seepage data, generally low salinity values in groundwater, and strong vertical salinity gradients (i.e., from low salinity in groundwater to higher salinity in porewater and surface water).
  - The arithmetic average TPAH (17) concentration in groundwater beneath this reach is the highest in the Study Area and approximately two orders of magnitude higher than that in shallow porewater.
  - The arithmetic average TPCB concentration in groundwater beneath this reach is the lowest in the Study Area and approximately one order of magnitude lower than that in shallow porewater.
  - Cu concentrations in groundwater are non-detect in most of the samples from this reach. The arithmetic average Cu concentration in groundwater is lower than that in shallow porewater.
- **Tributaries**
  - Groundwater discharges to the base of the Study Area in the tributaries. This conclusion is based on positive seepage data, generally low to moderate salinity in groundwater, and relatively strong vertical salinity gradients (i.e., from low salinity in groundwater to higher salinity in porewater and surface water); salinity in groundwater is greater on average in Dutch Kills and Whale Creek than in other tributaries, but is still lower than that in shallow porewater.
  - TPAH (17) concentrations in groundwater beneath the tributaries are generally lower than concentrations measured in groundwater beneath CM 0 – 1 and CM 2+, and are similar to concentrations measured in groundwater beneath CM 1 – 2. TPAH (17) concentrations in groundwater beneath the tributaries are similar to concentrations measured in shallow porewater, except for two locations in English Kills, where concentrations in groundwater are among the highest in the Study Area.
  - Arithmetic average TPCB concentrations in groundwater beneath all the tributaries are higher than those beneath the main stem of Newtown Creek. In addition, with the exception of English Kills, arithmetic average TPCB concentrations in

groundwater are similar to, or slightly higher than (within a factor of 3) those in shallow porewater. In English Kills, the arithmetic average shallow porewater TPCB concentration is higher than that in groundwater by a factor of 2.

- Similar to the main stem of Newtown Creek, Cu concentrations in groundwater beneath the tributaries are mostly non-detect (except in Dutch Kills) and are generally similar to concentrations measured in shallow porewater.

8.5 Sources

This section provides a summary of information regarding the sources of contaminants to the Study Area.

8.5.1 Historical Sources

Throughout the past 200 years, urban and industrial facilities present in the watershed of the Study Area have released contaminants to the creek. The current distribution of contaminants in the sediment column of the Study Area is due to uncontrolled historical and ongoing sources, historical dynamic fate and transport processes, and changes in contaminant loads over time. Therefore, the locations of impacts observed today cannot necessarily be directly linked to proximate upland sites or sources, including point sources. Notwithstanding this limitation, there is a great deal of relevant information available regarding both historical and current industrial operations and processes that may have released TPAH (17), TPCB, Cu, and other contaminants into the Study Area through various release and transport pathways. Historical sources are described in detail in Sections 3.2.6 and 3.2.8, as well as Appendix J. Current sources are summarized in Section 8.5.2.

8.5.2 Current Sources

Current (i.e., ongoing) sources to Newtown Creek are the focus of this CSM, due to the importance of controlling sources early in the RI/FS process as specified in the USEPA (2005a) contaminated sediment guidance. Chemicals may currently be entering the Study Area from multiple potential sources, including the following:

- Point sources and overland flow
- East River

- Groundwater
- Other sources
  - Bank erosion
  - Atmospheric deposition
  - Overwater activities
  - Shoreline seeps

Subsections 8.5.2.1 through 8.5.2.4 provide descriptions of these potential sources based on data collected during the RI, which are discussed in greater detail in Section 5 and Appendix E.

8.5.2.1 Point Sources and Overland Flow

As discussed in Sections 3.2.8.2 and 5.1, a variety of current point sources discharge to the Study Area. More than 300 private and municipal outfalls have been documented along Newtown Creek and its tributaries, some of which may be abandoned or no longer in use (see Figure 8-14). The point source discharges to the Study Area include the following:

- Individually permitted stormwater and wastewater discharges (i.e., treated effluent from groundwater remediation and dewatering systems, stormwater discharges from industrial sites, hydrostatic test water discharges, and discharges from secondary containment)
- Overflows from the combined sewer system and treated effluent overflow from the Newtown Creek WWTP
- Stormwater discharges (including point source discharges and overland flow discharges) from private, commercial, and industrial sites directly to the creek; from municipal infrastructure to the creek; or from a combination of both

The mass balances for TPAH (17), TPCB, and Cu (see Figures 8-15, 8-16, and 8-17, respectively; see also Section 6.5) provide an overall picture of the mass inventories and movement of contaminants by reach, including external loads that discharge into the defined Study Area boundary (e.g., point sources). Total point source annual contaminant loads are provided by discharge type and reach in Section 5.1.4.3.[294] Some notable findings regarding the point source loads are as follows:

- **CM 0 – 2**
  - Almost one-third of the point source TPAH (17) load (30% to 32%) to the Study Area is due to the effluent discharged from the Con Edison – 11th Street Conduit (DAR No. 110) dewatering system in CM 0 – 1.
  - Discharges in CM 0 – 2 contribute approximately 21% to 24% of the TPCB and 17% of the Cu point source loads to the Study Area, mainly through stormwater discharges.
- **CM 2+**
  - The smallest point source loads for TPAH (17), TPCB, and Cu to Newtown Creek occur in CM 2+. The TPAH (17), TPCB, and Cu loads represent 5% to 6%, 10% to 12%, and 9% of the total point source discharge to the Study Area, respectively.
- **Tributaries**
  - The majority of the point source TPAH (17) (51%), TPCB (65% to 67%), and Cu (74% to 75%) loads enter the Study Area in the tributaries, especially Maspeth Creek, East Branch, and English Kills.
  - In nearly all cases, the largest contributor of loads to the tributaries are CSOs, followed by stormwater (the only exceptions occur in the case of TPCB for some load estimation methods, in which stormwater contributes approximately equal to or greater loads than CSOs). The Newtown Creek WWTP treated effluent overflow, a large volume source that discharges to Whale Creek, contributes approximately 2% of the TPAH (17) load, 6% to 12% of the TPCB load, and 7% to 9% of the Cu load to the Study Area.

8.5.2.2 East River

The East River transports solids that contain contaminant concentrations consistent with the urban region as a load to the Study Area, due to the twice-daily tides. The tidal volumetric

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[294] The CFT model being developed to support the FS will be used to further evaluate these point source load estimates.

flow of East River water into Newtown Creek is approximately 70 times greater than the total point source volumetric discharges into Newtown Creek on an annual basis. When East River water flows into the creek during each flood tide, it mixes with the creek water, leaving behind particulate matter that deposits on the sediment bed. The East River is the primary source of the solids that deposit on the sediment bed in CM 0 – 2 (see Figure 8-8); these solids, along with upstream point sources, contribute to the solids that deposit in CM 2+. The East River is a significant source of solids to Dutch Kills and Whale Creek and contributes approximately 30% of the solids depositing in East Branch and English Kills on average (see Figure 8-8), with the contribution being higher in portions of these tributaries, especially the downstream ends. The East River contributes 15% of the solids deposited in Maspeth Creek.

The East River is strongly mixed by the tides, exhibiting an average tidal excursion of 22 kilometers, roughly, from The Battery to Rikers Island (Blumberg and Pritchard 1997). East River water mixes with the waters of upper New York Harbor and Long Island Sound, so it contains chemical constituents associated with those waterbodies.

Concentrations of TPAH (17), PCB, and Cu measured in East River surface water samples collected near the mouth of Newtown Creek are generally similar to those measured in CM 0 – 2 surface water during dry weather, reflecting the strong influence of the river on this reach of the Study Area. Contaminant loads from the East River to the Study Area reflect the difference between flood tide and ebb tide fluxes and are influenced by mixing within the water column and deposition rates. Estimation of these loads requires the use of linked hydrodynamic, sediment transport, and CFT models. This work is underway.

8.5.2.3 Groundwater

Groundwater is a potential ongoing source of contaminants to the Study Area. Groundwater discharge to the Study Area occurs at the base of the Study Area and through vertical permeable shorelines to the surface water (i.e., lateral discharge). Groundwater discharge to the base of the Study Area may provide chemical loads to subsurface sediment via transport into the interstitial spaces (as porewater) and sorption onto the solid matrix. Chemical transport from subsurface sediment to surface sediment and ultimately to surface water is discussed in relation to other fate and transport processes (see Sections 6 and 8.6). Chemical loading associated with lateral groundwater discharge is discussed in Section 8.6.1.1. In addition, as discussed in Section 8.6.1.3, groundwater contamination, where introduced into the base of the Study Area, is substantially attenuated in the subsurface sediment; this is evidenced by the large reduction of concentrations between groundwater and porewater just beneath the surface sediment (15 to 30 cm [6 to 12 inches] depth). As discussed previously and further in Section 8.6.1.3, porewater concentrations can be impacted by tidal exchanges. Although there are no direct contaminant measures associated with tidal exchange, the lines of evidence presented in Section 6.4.3.1.2 indicate that this process is not a primary driver of shallow porewater concentrations. The following discussions relate to groundwater discharge to the base of the Study Area.

Groundwater within and adjacent to the Study Area occurs in three water-bearing units—fill, post-glacial deposits and the UGA. The UGA is the most transmissive water bearing unit among the three and is present under either unconfined or semi-confined conditions. Although the postglacial deposits and fill units also transmit groundwater, primarily under unconfined conditions, these units are generally less transmissive than the UGA and are laterally discontinuous. The UGA is continuous across the Study Area and is the primary source of groundwater discharge to the base of the Study Area. Groundwater within the fill and post glacial deposits discharges directly to the surface water through vertical permeable bulkheads. However, where it encounters low permeability bulkheads, groundwater flows under or around these barriers and discharges to the base of the Study Area. The UGA is concluded to be the primary source of groundwater to the Study Area; this conclusion may be revised in the future based on data collected as part of the planned USEPA study of lateral groundwater discharge in the shallow fill.

Groundwater that migrates into the subsurface sediment and then as porewater through the subsurface and surface sediment ultimately reaches the surface water. The groundwater flow rate, estimated using seepage rate measurements, varies spatially throughout the Study Area (see Figure 8-18 and Section 5.2), as follows:

- **CM 0 – 2**. Areas with positive flow (discharge into the Study Area) and negative flow (net loss from the creek to groundwater) are observed in CM 0 – 2. Generally, net positive flow is observed in CM 0 – 1, whereas net negative flow is observed in CM 1 – 2, which is attributed to groundwater extraction and treatment.

- **CM 2+**. Net positive flow is observed upstream from CM 2 in the main stem. The highest positive flow rates occur in the southeastern portion of the Turning Basin near CM 2.6.
- **Tributaries**. Net positive flow, indicating net groundwater discharge to the base of the Study Area, is observed in all tributaries. Groundwater discharge rates are generally higher at or near the heads of the tributaries (except for Maspeth Creek), with the highest groundwater discharge rate in the tributaries observed near the head of English Kills.

Groundwater contaminant loads to subsurface sediments,[295] as well as chemical mass inventories and loads by reach, are presented in the mass balance (see Figures 8-15, 8-16, and 8-17 and Section 6.5). These figures illustrate a key consideration in the evaluation of groundwater in this RI: groundwater contaminant loads to subsurface sediment do not translate directly to contaminant loads to surface sediment and surface water, due to the extensive attenuation in the subsurface sediment and partitioning processes. For example, the total groundwater TPAH (17) load to the base of CM 2+ is estimated to be between 740 and 1,400 kg/year (see Figure 8-14), whereas the total porewater load of TPAH (17) from subsurface sediment to surface sediment in CM 2+ is approximately 100 to 200 times less (7.3 kg/year). This attenuation is discussed further in Section 8.6 and will be further evaluated using the CFT model that is being developed to support the FS.

Other notable findings regarding chemical loads from groundwater into subsurface sediment are as follows:

- Groundwater loads of TPAH (17) and TPCB to subsurface sediment in CM 0 – 2 are minor compared to those in the rest of the Study Area. As a percentage of Study Area totals, groundwater chemical loads to CM 0 – 2 are 7% to 10% (TPAH [17]) and 2% to 4% (TPCB).

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[295] As discussed in Section 4.9.3, due to the uncertainty in the partition coefficients for TPAH (17) and TPCB, dissolved phase concentrations were estimated using two methods (one that uses site-specific dry weight partition coefficients and one that uses literature-based OC partition coefficients), which resulted in the development of a range of estimated groundwater loads to the Study Area (see Sections 5.2.2 and 6.1 of Appendix F).

- Approximately 90% of the TPAH (17) groundwater load to subsurface sediment occurs in CM 2+.[296]
- A large percentage (82% to 89%) of the groundwater load of TPCB into subsurface sediment in the entire Study Area occurs in English Kills.
- Cu loads to subsurface sediment are generally evenly distributed between the main stem of Newtown Creek and English Kills, with more than 90% of the Cu load discharging to these two reaches. The combined Cu loads from groundwater to subsurface sediment in East Branch, Maspeth Creek, Dutch Kills, and Whale Creek are less than 10% of the total for the Study Area.

Lateral groundwater discharge through vertical permeable shorelines also may transport contaminants to the water column. To evaluate the potential for lateral groundwater discharge to influence surface water quality within the Study Area in this RI Report, dry weather surface water data adjacent to the five areas with the highest estimated lateral groundwater discharge rates per linear foot of shoreline were compared to those in surrounding areas of the creek (see Section 6.4 of Appendix F). The comparison indicated no observable influence from lateral groundwater discharge on surface water chemical concentrations. However, loads from lateral groundwater discharge will be further evaluated during the FS with the CFT model (through sensitivity analysis) and with data (if available) from the USEPA study.

8.5.2.4 Other Sources

Shoreline erosion, atmospheric deposition, overwater activities, and shoreline seeps are minor sources of contaminants to the surface water and surface sediment of Newtown Creek (see Section 5), as shown in the following:

- **Shoreline erosion**. TPAH (17), TPCB, and Cu concentrations from surface sediment samples collected on or near potentially erodible shorelines are generally within the range of, or lower than, concentrations of the rest of the RI surface sediments collected in the same general area (see Figures 5-39 through 5-42). However, some sample locations exhibit higher concentrations. This indicates that, in general, shoreline

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[296] This value may be an overestimate due to the potential for NAPL entrainment in the samples (see Section 4.9.3.1; see also Section 6 of Appendix F).

erosion is likely not a significant ongoing source of contamination to the Study Area. Potentially significant ongoing shoreline sources are discussed in Section 5.4.4.

- **Atmospheric deposition**. The estimated atmospheric deposition loads for TPAH (17), TPCB, and Cu (reported in a study performed by the NJADN [Reinfelder et al. 2004]) are 21 to 170 times lower than loads estimated from point sources and groundwater, depending on the specific chemical (based on a comparison to the higher of the two values; see Table 5-29).
- **Overwater activities**. As described in Section 5.6, overwater activities are not considered a significant ongoing source to the Study Area, based on the minimal record of releases associated with overwater activities (in the NYSDEC Spill Incidents Database), and with the current regulations and BMPs in place. However, they may have been significant historically. Historical overwater activities are summarized in Section 3.2.11. The potential for such activities to result in a future significant release exists, due to the nature of the industrial activities that are expected to continue well into the future, given the industrial nature of the waterway.
- **Shoreline seeps**. Seeps from shorelines can represent an ongoing source of contaminants to the Study Area, as discussed in Section 5.7. Intermittent shoreline seeps have been observed at discrete locations in the Study Area (including observations of bulkhead NAPL seeps by others—see Section 5.7.3). TPAH (17), TPCB, and Cu concentrations from opportunistic seep samples collected during FS Part 1 field activities are generally within the range of, or lower than, dry weather surface water samples from the Phase 1 and Phase 2 field activities (see Figures 5-46 through 5-49). This suggests that most of the opportunistic seep samples collected likely represent bank storage (i.e., water that flows into interstitial spaces on an incoming tide and seeps out of the face of the bank as the tide goes out). As discussed in Section 8.5.2.3, lateral groundwater discharge through vertical permeable shorelines also may transport contaminants to the water column. Since there is no observable influence from lateral groundwater discharge on surface water chemical concentrations, such loads are highly localized and are likely not significant to the overall Study Area. However, this conclusion may be revisited in the future; because shallow lateral groundwater discharge inputs to Newtown Creek have not been empirically characterized, USEPA is planning a study to collect empirical data to support the FS and reduce uncertainty in the current lateral groundwater discharge estimates.

8.6 Fate and Transport

8.6.1 Contaminant Fate and Transport Processes

To support CERCLA decision-making, one of the key objectives of this RI is to describe ongoing significant sources of contaminants to the surface water and sediment that must be controlled to effectively address potential risks to human health and the environment. Because impacts to human health and the environment stem from contact with and uptake of contaminants by humans and ecological receptors, a central objective of the RI is to understand the sources of these contaminants. In a system such as Newtown Creek, the two most important sources to receptors, either through direct contact and/or dietary exposure, are contaminants in surface sediment and surface sediment porewater, and contaminants in the water column. These are the primary sources of contamination to the food web in which bioaccumulation and/or biomagnification can occur. This section focuses on characterizing sources of contaminant loads to surface sediment and the water column, as well as the fate and transport processes affecting them. Bioaccumulation is discussed in Section 8.7.

Sections 8.6.1.1 through 8.6.1.4 discuss the following, with respect to contaminant fate and transport:

- Sources of contaminants to the water column
- Fate and transport processes affecting contaminants in the water column
- Fate and transport processes in subsurface sediment, with a focus on processes that can transport contaminants from subsurface sediment to surface sediment and/or surface water, such as groundwater flow
- Surface sediment fate and transport processes, including exchange between surface sediment and surface water, for both particulate and dissolved phase contaminants

8.6.1.1 Sources to the Water Column

Point source discharges and overland flow, lateral groundwater flow, discharge of surface porewater (driven by groundwater flow), and East River tidal waters enter the Study Area water column, where contaminants and solids are transported throughout the creek. The importance of these sources differs by flow regime (wet versus dry weather) and tide (incoming versus outgoing). During dry weather conditions, tidal exchange with East River water strongly influences the concentrations and transport of contaminants in surface water in the main stem and lower tributaries of the Study Area. In the upper reaches of the Study Area, it is likely that the flux from surface sediment porewater contributes to the generally higher contaminant concentrations reported in surface water during dry weather conditions, compared with those observed in the downstream reaches (and the East River; see Section 6.4.2). Additionally, as discussed in Section 8.5.2.3, contaminants associated with lateral discharge of groundwater through vertical permeable shorelines and shoreline seeps including NAPL seeps may also affect dry weather surface water concentrations. As noted in Section 8.5.2.3, these discharges do not appear to influence dry weather surface water concentrations over large spatial scales, but this process is being further evaluated as part of the FS with the CFT model and through the empirical data collection planned by USEPA.

Under wet weather conditions, point source discharges and overland flow are a source of elevated contaminant concentrations in the water column throughout much of the Study Area, with higher contaminant concentrations observed in CM 2+ and the tributaries than in CM 0 – 2. Contaminants associated with these point source discharges and overland flow do not flush out of the Study Area over a single tidal cycle; instead, tracer simulations and sampling data indicate it typically takes several tidal cycles over the course of a few days to return to surface water contaminant levels typically observed under dry weather conditions (see Section 6.2.2). Contaminant concentrations in surface water during wet weather may also be affected by localized sediment erosion in some locations, especially in the areas where the inflows are higher and the water is very shallow.

Additional evaluation and quantification of these sources and loadings to the water column, including comparisons under different flow regimes, can be progressed using the CFT model that is being developed to support the FS.

8.6.1.2 Fate and Transport Within the Water Column

As noted in Section 8.6.1.1, the contaminant mass discharged into the water column following a wet weather event remains within the Study Area for at least several tidal cycles. During this time, some contaminants disperse from their sources, partition between dissolved and particulate phases, and settle onto the sediment bed. In addition, some contaminants (e.g., TPCB and TPAH [17]) may volatilize into the atmosphere to a limited degree (see Section 6.4.6).

Because many contaminants, including TPAH (17), TPCB, and Cu, sorb strongly to particles in the water, their transport and dispersal depends (in part) on the settling properties of the particulate matter. Larger particles settle closer to the release point, and finer particles are generally transported farther. In general, contaminants sorb more strongly to finer particles than to particles that are coarser. This process contributes to the observed distribution of contaminants in surface sediment (see Section 4.2.3)—the generally observed increase in contaminant concentrations with distance downstream from the head of the tributaries is likely due (in part) to differential settling of fine and coarse contaminated particulate matter.

8.6.1.3 Subsurface Sediment Fate and Transport Processes

Bed properties and stability. The subsurface sediment of Newtown Creek is highly organic. TOC contents in both the surface and subsurface sediment are high, especially in CM 2+ and the tributaries. The sediment boundary with the underlying native material, which is low in organic matter, is generally well defined. In the upstream tributaries, the subsurface sediment is composed predominantly of solids discharged from point sources (see Figure 8-8). In CM 2+, the subsurface sediment is a mixture of point source and East River solids, and in CM 0 – 2 the composition contains a greater contribution of East River solids compared to the upstream reaches.

The subsurface sediment bed is generally physically stable (i.e., net erosion that exposes subsurface sediment and deeper depths is not occurring in meaningfully large areas), as indicated by the following:

- Lower concentrations in surface sediment as compared to subsurface sediment at 80% of the sampled locations in the Study Area (see Section 4.3.4)
- Low current velocities throughout the Study Area that result in minimal or no erosion of the sediment bed (see Sections 4 and 5 of Appendix G)
- Pre- and post-Hurricane Sandy bathymetry surveys, which indicate minimal erosion of the sediment bed, even during anomalous current velocities generated by

the storm surge (see data in Appendix C7 of Phase 1 DSR Submittal No. 3, within Attachment Bi-A of Appendix Bi)

The deposition of solids originating from outside the Study Area (primarily from the East River, point sources, and overland runoff) results in burial within the existing sediment bed. This accretion of newly deposited solids on the surface of the bed contributes to bed stability and also reduces the availability to the food web of contaminants that are currently in the surface sediment. Due to mixing processes, the biologically active zone contains a combination of newly deposited and historically deposited solids and contaminants. Over time, as deposition proceeds, historically deposited solids and contaminants are continually covered up and diluted by newly deposited materials, so the contribution of historically contaminated materials to the food web declines. Mixing processes in the surface sediment include propwash resuspension and bioturbation, which are largely limited to the surface layer, along with other processes that may result in localized deeper mixing or disturbances, such as maintenance dredging, bridge and bulkhead maintenance, and bulkhead and shoreline collapse. Mixing processes are evaluated further as part of the sediment transport and CFT modeling.

Groundwater and porewater flow and contaminant transport. Dissolved and free phase (i.e., sheen or NAPL) contaminants can be transported within subsurface sediment by the processes of porewater flow and gas ebullition, both of which have been investigated as part of this RI. With the exception of the areas noted in the list that follows, groundwater flows from the native material into the subsurface sediment of the Study Area. Groundwater within the subsurface sediment (subsurface porewater) moves upward into the surface sediment and mixes with surface sediment porewater, which then undergoes exchanges with the overlying water column (see next subsection). In the following sections of the Study Area, the net flow is from sediment into native material (i.e., negative) (see Section 5.2):

- Just downstream of the Pulaski Bridge near CM 0.5
- Just upstream of the mouth of Dutch Kills near CM 1.1
- The central portion of Newtown Creek between CM 1.5 and 2.0

As soon as groundwater from the native material enters the subsurface sediment, it is referred to as porewater for the purposes of this RI. Where the creek is gaining (i.e., flow is from native material into the sediment), this porewater flow passes through the subsurface sediment bed and into the surface sediment. As discussed in Section 6.4.5, groundwater contamination, where present, is substantially attenuated by the subsurface sediment, as evidenced by the large reduction in reach-average concentrations between groundwater and porewater just beneath the surface sediment (15 to 30 cm [6 to 12 inches] depth). Contaminants in porewater equilibrate with sediment—this is evidenced by strong relationships that have been observed between contaminant concentrations in sediment and collocated porewater; these site-specific data show partitioning of contaminants to sediment is strong, but the process is complex and not entirely controlled by OC or soot carbon, unlike many other sites (see Section 6.4.1). This means that the contaminant concentrations in porewater throughout the subsurface sediment column reflect equilibration with the collocated particulate phase material. Therefore, the loads of contaminants entering the surface sediment due to porewater flow from the shallow subsurface sediment below are determined by two factors—the advective porewater flow rate and the concentrations of contaminants in the shallow subsurface sediment (i.e., the subsurface sediment lying just below the surface sediment (characterized by samples collected from the 15- to 30-cm [6- to 12-inch] depth interval).[297] Thus, due to attenuation resulting from sorption processes (as well as dispersion and possibly degradation), these loads are not determined by deeper, often higher concentrations in the subsurface sediment, nor are they determined by contaminant concentrations in groundwater.

Contaminants in the groundwater that contribute a chemical load to subsurface sediment (see Section 5.2) are attenuated significantly by these sorption and dispersion processes. PAHs are subject to microbial degradation as well (see Section 6.4.6). Subsurface sediment in the Study Area averages approximately 10 feet thick, and the residence time of the groundwater that flows through the subsurface sediment ranges from days to months (see Section 6.4.5.3), although the chemicals move at a much slower rate due to retardation from the process of sorption. These residence times are long enough that the dissolved chemicals in the groundwater flowing upward from the native material can exchange with

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[297] As discussed previously, shallow porewater concentrations and loads can be impacted by tidal exchanges in which surface water can enter the surficial sediments during periods of higher tides. Although there are no direct contaminant measures associated with tidal exchange, the lines of evidence presented in Section 6.4.3.1.2 indicate that this process is not a primary driver of shallow porewater concentrations (both 0- to 15-cm and 15- to 30-cm depths).

(and equilibrate with) the chemicals bound to the solid phase in the subsurface sediment layer. Thus, the subsurface sediment effectively sorbs the majority of the contaminants, so that dissolved contaminants coming from the native material are retained by and attenuated within the subsurface sediment bed (by sorption, dispersion, and degradation), rather than being transported directly to the surface sediment and surface water.

This attenuation is quantified in the mass balance (see Figures 8-15, 8-16, and 8-17 and Section 6.5). In the case of TPAH (17), in which the load from groundwater is relatively large compared to other source terms, on a Study Area-wide basis, the annual porewater load of TPAH (17) flowing upward into the surface sediment from the subsurface sediment (based on measured porewater concentrations at the 15- to 30-cm [6- to 12-inch] depth interval) is 20 kg/year. That mass load is only 1 to 2% of the load from groundwater flowing to subsurface sediment (830 to 1,500 kg/year) from the native material, which illustrates the large degree of attenuation by the subsurface sediment. Furthermore, the annual groundwater TPAH (17) load entering the subsurface sediment represents a relatively small portion of the subsurface sediment TPAH (17) inventory (approximately 0.1%). In the case of both TPCB and Cu, the groundwater load is among the smallest mass transport terms. Therefore, the process of attenuation from sorption to surrounding sediment is comparatively less important for these chemicals.

In addition, contaminants dissolved in the groundwater provide only a minimal contribution to the much larger mass of contaminants currently present in the stable subsurface sediment. As described in the mass balance (see Figures 8-15, 8-16, and 8-17 and Section 6.5), the mass of the key contaminants entering the base of the Study Area sediments via groundwater annually is as follows:

- **TPAH (17)**. Approximately 830 to 1,500 kg, which is approximately 0.1% of the mass of TPAH (17) in subsurface sediment (1,300,000 kg).
- **TPCB**. The annual groundwater load is 0.045 to 0.27 kg, which is less than 0.002% of the subsurface sediment mass of 16,000 kg.
- **Cu**. The annual groundwater load is 10 kg, which is less than 0.001% of the subsurface sediment mass of 4,000,000 kg.

These differences illustrate the substantial sorptive and attenuation capacity of the subsurface sediment.

The effects of groundwater loadings on subsurface and surface sediment will be further evaluated and quantified using the CFT model that is being developed to support the FS.

Gas ebullition. Gas ebullition originates primarily in surface and shallow subsurface sediment, when water/sediment temperatures are highest and when water depths are shallowest (near the hours of lowest tides). Once formed, the gas bubbles may migrate upward in the sediment, depending on the overlying sediment strength and hydrostatic pressure, and be released to the water column. Gas ebullition requires a relatively labile source of organic matter, which in Newtown Creek originates primarily from CSO and storm drain discharges (e.g., fecal matter), and potentially other organic contaminants (e.g., sheen and NAPL). Factors favorable for gas ebullition include sediment that has low strength, low permeability, and high organic content; warm sediment temperatures; and shallow water (and therefore, low hydrostatic pressure). When gas ebullition occurs in the presence of sheen-bearing material (NAPL or other contaminants), those constituents may be transported with gas bubbles to the water column. Areas in Newtown Creek where conditions are favorable for gas ebullition, and where this transport mechanism has been documented, predominantly include portions of Dutch Kills, East Branch, English Kills, and the Turning Basin. Based upon the RI field investigation surveys, gas ebullition does not appear to be as prominent in CM 0 – 2, potentially because of increased water depth and/or lower organic content in the sediments there.

Additional discussion of the gas ebullition process and the results of some preliminary studies of ebullition in the Study Area are provided in Section 6.4.7 of this RI Report and Sections 2, 3, and 5 of Appendix D. Quantitative ebullition testing was performed as part of the FS Part 2 field program (data for the FS Part 2 field program are not included in the RI Report and are presented in the FS Gas Ebullition DER [Anchor QEA 2022b]).

NAPL advection in sediment and native material. At NAPL saturations above the residual saturation, NAPL is continuously connected between pore spaces and has the potential to flow, or advect, if the driving forces acting on the NAPL (i.e., hydraulic and gravitational) are sufficient to overcome the capillary forces that resist NAPL movement. The NAPL mobility evaluation (using centrifuge testing) performed on subsurface sediment and native material in CM 0 – 2 showed that NAPL in CM 0 – 2 subsurface sediment and native material, where present, is immobile. Quantitative NAPL mobility testing for CM 2+ and the tributaries was performed as part of the FS Part 2 field program (data for the FS Part 2 field program are not included in the RI Report and are presented in the FS NAPL DER [Anchor QEA 2022a]).

8.6.1.4 Processes Associated with Surface Sediment

The fate and transport processes associated with the surface sediment are described in this subsection (see Figure 8-1). Solids deposited from point sources, particularly CSOs and stormwater, and from the East River become mixed within the shallow surface sediment layer via biological and physical processes (including propwash, as described later in this subsection), resulting in contaminant concentrations in the near-surface sediment that are a blend of current and historical contaminant loads (as introduced in Section 8.6.1.3). Through partitioning processes, other ongoing sources of contaminants to the Study Area surface water (including lateral groundwater discharge, shoreline seeps, atmospheric deposition, shoreline erosion, and overwater activities) and fluxes from sediment to surface water (from porewater advection, tidal exchange, and gas ebullition) contribute to the current contaminant loads associated with these depositing solids, to varying degrees and depending on location in the Study Area.

In the CM 0 – 2 main stem, where NSRs are the highest and solids deposition is most strongly influenced by East River tidal exchange, the blended surface sediment concentrations are within the range of particulate phase concentrations measured in East River surface water samples (see Section 6.4.3.3). Although the NSRs in CM 0 – 2 are most strongly influenced by East River tidal exchange, other processes and sources of contaminants to the surface water in the reach can contribute to the chemical concentration of those solids as they deposit, including groundwater discharge, propwash resuspension, contaminant exchange with CM 2+, point sources, and shoreline seeps. The relative magnitude of influence of each of these processes will be evaluated during the FS CFT modeling. In CM 2+, where NSRs are relatively low and solids deposition is influenced both by CSO and stormwater discharges and East River tidal exchange, contaminant concentrations are higher, reflecting both the ongoing and historical contributions. In the tributaries, where NSRs are variable and solids deposition is dominated by CSO and stormwater discharges (Maspeth Creek, English Kills, and East Branch) or East River (Dutch Kills and Whale Creek), contaminant concentrations also reflect both the ongoing and historical contributions. Some of these tributaries, mainly English Kills and East Branch, exhibit spatial gradients in surface sediment concentrations in which the upstream portions of these tributaries more closely resemble the lower concentrations of the current point source discharges, whereas the higher concentrations in downstream portions of the tributaries reflect the higher-level historical discharges and releases (see Sections 3.2.6 and 3.2.8.1; see also Section 4.2.4 of Appendix E for more information). These patterns likely reflect a combination of differential settling of coarser versus finer solids containing differing contaminant concentrations, as well as blending of current and historical sediment and influences of other ongoing sources. Because surface sediment concentrations in the tributaries are generally higher than those of current point sources, this difference indicates mixing between newly depositing sediment and higher concentration historical sediment is ongoing and/or that there are contributions from other ongoing sources, such as those listed in the previous paragraph.

Contaminants can be transported from the surface sediment to the surface water in particulate form, due to propwash resuspension, storm event erosion (in localized shallow areas in certain tributaries), biological processes (see Section 6.4.3), or facilitated by gas ebullition. Surface water can also enter the surficial sediments during periods of higher tides, as evidenced by the porewater seepage flow reversals captured by the USGS seepage studies (see Attachments Bi-B9 and Bii-B1 of Appendices Bi and Bii, respectively). Under typical conditions, there is minimal or no erosion of the sediment bed due to tidal forces. Based on an analysis of bathymetric data collected pre- and post-Hurricane Sandy, only minor bed scour occurs, even during a hurricane's storm surge (see data in Appendix C7 of Phase 1 DSR Submittal No. 3, within Attachment Bi-A of Appendix Bi). Under storm conditions, there may be limited sediment erosion in some locations, especially in the areas where the inflows are higher and the water is very shallow (i.e., at the heads of the tributaries). Propwash from ship traffic causes episodic bed scour within the navigation channel and Turning Basin, with subsequent redeposition and dispersal of solids (see Section 5.3 of Appendix G). As a result of these processes, contaminants are resuspended into the water column. Once in surface water, contaminants partition to and from particulate matter and can then be transported to other reaches within the Study Area or out of the Study Area over the tidal cycle, or redeposited on the sediment bed within the same reach or a different reach of the Study Area. In the case of propwash resuspension, modeling (see Sections 5.3 and 5.5.3 of Appendix G) indicates the majority of the resuspended sediments settle in the same general area from where they were disturbed, which has the net effect of a shallow mixing process. Indeed, data from sediment traps indicate depositing material sampled in the water column represents a combination of sediment from external sources (e.g., East River, point sources) and local surface sediment.

Dissolved phase contaminants are present in the surface sediment porewater. Processes that influence surface sediment porewater concentrations, which include processes that originate from beneath the surface sediment (i.e., porewater advection) and processes that originate from the overlying surface water (i.e., tidal exchange), were evaluated to assess their relative importance when compared with partitioning with the surface sediments (see Section 6.4.3.1.2). Based on the results of the analyses (which included multiple lines of empirical evidence), it was concluded that surface porewater (dissolved phase) chemical concentrations are largely controlled by partitioning with sediments (sorption/desorption effects) and that tidal exchange and porewater advection have, at most, secondary impacts on the observed concentrations. However, because surface porewater contaminant concentrations are higher relative to surface water concentrations for most contaminants, dissolved phase transport of contaminants to the surface water occurs through a combination of diffusion, porewater advection due to groundwater flow, and tidal exchange. Contaminants in the Study Area sorb strongly to sediment, although they range in the degree of sorption by several orders of magnitude (see Section 6.4.1), and there are differences among reaches in some cases. Chemicals that are less sorptive are present in porewater at relatively higher concentrations and result in larger dissolved phase contaminant transport, whereas chemicals that are more sorptive tend to be more strongly bound to the solid phase and have lower dissolved phase contaminant transport (see Section 6.4.1).

For chemicals with strong sorption, the effects from deposition of particulate phase mass (which acts to transport contaminants from surface sediment to subsurface sediment by burial) are typically much greater than the effects of upward advection of dissolved phase mass driven by groundwater seepage (see Section 6.4.3). As such, porewater advection is relatively more significant only in areas with lower relative NSRs and higher seepage rates, and primarily for the most mobile contaminants (e.g., LPAHs).

All of the previously mentioned processes, which affect surface sediment solid phase and dissolved phase concentrations, also control the extent to which contaminants are bioavailable to the food web. Bioaccumulation is described in Section 8.7.

8.6.2 Natural Recovery

Natural recovery refers to the process by which chemical concentrations, primarily in surface sediment, decline over time without specific intervention designed for that purpose. Natural and anthropogenic processes that can reduce chemical concentrations in aquatic sediment include the following:

- Chemical or biological processes that convert contaminants to less toxic forms (e.g., biodegradation)
- Physical processes that reduce surface sediment contaminant concentrations through mixing or burial (e.g., sedimentation)
- Physical and/or biological processes that result in the loss of contaminants to surface water (e.g., groundwater advection, tidal exchange, bioturbation, gas ebullition)
- Reductions in contaminant loads to the system, which over the long term will act to reduce surface sediment concentrations and reduce the effects of sources internal to the system, such as flux from surface sediment to surface water

Natural recovery through burial is ongoing throughout the entire Study Area, as evidenced by the observation that the concentrations of TPAH (17), TPCB, and Cu in nearly all Phase 1 and Phase 2 sediment cores in the surface sediment segment (0- to 15-cm [0- to 6-inch] depth interval) are lower than the concentration in the next deepest subsurface sediment core segment (approximately 15- to 60-cm [6- to 24-inch] depth interval) (see Section 4.3.4.1 and Figures 4-93, 4-96, and 4-98). Natural recovery by burial is further evidenced by the high-resolution cores, many of which indicate that concentrations increase with depth even within the top few cm of the sediment surface (see Figure 8-19, which shows an example of such a core in which the estimated NSR is 0.4 cm/year; all high-resolution cores are shown in Figures 4-92a through 4-92d).

Evidence of natural recovery is also supported by the mass balance (see Figures 8-15, 8-16, and 8-17 and Section 6.5). The mass balance provides an overall picture of the mass inventory and movement of contaminants through the system, including external loads to the Study Area (from groundwater, point sources, East River, and the atmosphere), as well as loads among the media of the Study Area. Many of the components of the mass balance have been quantified and integrated into discussions in this CSM (some others have not yet been quantified, such as gas ebullition, lateral groundwater discharge, and contaminant loadings from the East River, which will be evaluated during the FS). The mass balance around surface sediment for TPAH (17), totaled over the Study Area, indicates that natural recovery is occurring. Sources to surface sediment total 40 to 70 kg/year (equal to the sum of deposition flux [20 to 50 kg/year] and subsurface porewater advection [20 kg/year]). Losses total 330 kg/year (equal to the sum of porewater diffusion and tidal exchange [80 kg/year], porewater advection [13 kg/year], and burial [240 kg/year]). Thus, there is a net loss of 260 to 290 kg/year (equal to the difference between the sources [40 to 70 kg/year] and the losses [330 kg/year]); so, natural recovery is ongoing. This loss can be compared with the current mass of TPAH (17) in surface sediment (3,400 kg), suggesting a current recovery rate of 8% to 9% per year. According to the mass balance, the rate of recovery is fastest in CM 0 – 2, where the NSRs are the greatest and where burial of surface sediment is the fastest. Moreover, concentrations of TPAH (17) in sediment are likely declining due to biological degradation (see Section 6.4.6), which likely occurs both in surface and subsurface sediment (at differing rates).

Similarly, when the mass terms for TPCB and Cu are totaled over the Study Area surface sediment, the differences in loss and source terms also show evidence of natural recovery. For TPCB, losses from surface sediment [20 kg/year] exceed sources [1 to 3 kg/year], suggesting a net loss of 17 to 19 kg/year and a current recovery rate of 5% to 6% per year (the current mass of TPCB in the surface sediment is 330 kg). For Cu, losses from surface sediment [2,800 kg/year] exceed sources [700 to 1,000 kg/year], suggesting a net loss of 1,800 to 2,100 kg/year and a current recovery rate of 3% to 4% (the current mass of Cu in the surface sediment is 58,000 kg). Thus, for all three chemicals, the RI data indicate that natural recovery is ongoing.

The rate of natural recovery will be further evaluated with the CFT model as part of the FS.

Finally, contaminant concentrations in surface sediment have also declined over time, due in part to declines in contaminant loads to the Study Area. There are two major factors leading to the decrease in contaminant loads in the Study Area:

1. A decrease in industrial sources, due to the changing nature of industrial operations along Newtown Creek and remediation of upland sites with legacy contamination
2. Improvements in wastewater and stormwater management within the Newtown Creek sewershed, as well as regionally, over time. This includes upgrades to the WWTPs and other discharge management and infrastructure changes (e.g., BMPs, construction of combined sewer interceptors, permitting of industrial discharge connections to sewers, limiting of industrial discharges to combined sewers during wet weather, and control of unauthorized discharges)

Although there has been a decline in loads from these sources, there are still loads due to the volume of discharge occurring in CSO and stormwater releases and the urban contamination associated with these flows, as well as due to releases from potentially impacted upland sites (see Appendix J) and other ongoing sources such as lateral groundwater discharge, shoreline seeps, atmospheric deposition, shoreline erosion, and overwater activities.
While the RI data indicate that natural recovery is occurring throughout Newtown Creek, episodic events can impact natural recovery on a local basis. Subsurface sediment concentrations may be exposed due to anthropogenic disturbances, which can slow natural recovery temporarily and locally. These resuspended subsurface sediments may then deposit in the same location or elsewhere within the Study Area, slowing recovery.

Natural recovery is anticipated to continue in the future, due to continued sediment deposition, further isolating subsurface sediment contaminants from exposure to the food web. Future improvements in stormwater and CSO management and treatment may also lead to further declines in contaminant loads from point and non-point sources, resulting in further declines in surface sediment concentrations.

Finally, natural recovery can be incorporated into a risk reduction program for contaminated sediment, relying on ongoing, naturally occurring processes that contain, destroy, or reduce the bioavailability or toxicity of contaminants in sediment (USEPA 2005a). Because surface sediment is an important potential exposure source for the aquatic food web, decreases in surface sediment concentrations over time are a key metric used to evaluate natural recovery processes. Changes in surface sediment concentrations will be evaluated as part of future CFT modeling efforts.

8.7 Bioaccumulation, Risk, and Exposure Pathways

The finalized BHHRA and BERA are included in Appendices H and I, respectively, and are summarized in Section 7. The risk assessments evaluated the potential human health and ecological risks that will provide one set of criteria used during selection of a CERCLA remedy in the FS.

Based on the current and future uses of the Study Area, the following exposure scenarios were identified for quantification of risks to human health in the BHHRA: five categories of recreational users, four categories of occupational users, one category of unauthorized users (sailboat users), and one general exposure scenario involving residents and occupational workers (local flooding scenario).

- Recreational users
  - Boaters/swimmers: assessed potential dermal contact with and incidental ingestion of surface water and inhalation of ambient air exposure pathways. The dermal contact and incidental ingestion of surface sediment were qualitatively evaluated in the BHHRA, as these exposure pathways are considered complete but with low exposure potential.
  - Recreational anglers and crabbers: assessed potential dermal contact with surface water, inhalation of ambient air, and the ingestion of fish and crab tissues exposure pathways. Incidental ingestion of surface water was qualitatively evaluated in the BHHRA, as this exposure pathway is considered complete but with low exposure potential.
  - Plank Road Area recreational users: assessed potential dermal contact with surface water, dermal contact and incidental ingestion of surface sediment, and inhalation of ambient air exposure pathways (USEPA 2014c). The incidental ingestion of surface water was qualitatively evaluated in the BHHRA, as this exposure pathway is considered complete but with low exposure potential.

  - Shoreline recreational users: assessed potential inhalation of ambient air exposure pathway.
- Occupational users
  - Landside workers: assessed potential inhalation of ambient air exposure pathway.
  - Dockside workers: assessed potential dermal contact with and incidental ingestion of surface water, dermal contact and incidental ingestion of surface sediment, and inhalation of ambient air exposure pathways.
  - Future Hunter's Point South construction workers: assessed potential dermal contact with and incidental ingestion of surface water, dermal contact and incidental ingestion of surface sediment, and inhalation of ambient air exposure pathways (USEPA 2014d).
  - General construction workers: assessed potential dermal contact with and incidental ingestion of surface water, dermal contact and incidental ingestion of surface sediment, and inhalation of ambient air exposure pathways (USEPA 2014d).
  - The BHHRA estimates risks posed by a site in the absence of remediation or exposure controls. Thus, the BHHRA does not assume the use of worker protections and criteria regarding chemical contamination that might be required under health and safety regulations.
- Unauthorized users
  - Sailboat users: assessed potential dermal contact with surface water, dermal contact and incidental ingestion of soil/fill, and inhalation of ambient air exposure pathways (USEPA 2014d). Surface sediment samples collected adjacent to the areas where the sailboats are moored serve as a surrogate to represent the soil/fill material. The incidental ingestion of surface water was qualitatively evaluated in the BHHRA, as this exposure pathway is considered complete but with low exposure potential. As described in Section 4.4.8.10 of the BHHRA, these sailboat users are visitors to illegally moored sailboats along the bulkheads near Vernon Boulevard in the Study Area (see Appendix H).
  - Trespassers/homeless people: qualitative evaluation.

- Flooding scenario
  - Residents and occupational workers: assessed potential dermal contact with and incidental ingestion of surface water and dermal contact and incidental ingestion of surface sediment exposure pathways (USEPA 2014d). Inhalation of ambient air during flooding would also occur and was evaluated qualitatively, due to the uncertainty in estimating air concentrations related to flooding events.

The receptor categories and exposure pathways for these receptors are presented in the BHHRA exposure pathway CSM figures for current and future conditions (see Figures 7-1 and 7-2, respectively). The conclusion of the BHHRA risk characterization is that the only unacceptable human health cancer risks or noncancer hazards were associated with recreational fish and crab consumption, and general construction work.

The aquatic organisms and semiaquatic wildlife (the receptors) evaluated in the BERA were selected based on the outcome of surveys conducted in the Study Area in Phase 1 and subsequent discussions with USEPA during development of the *Baseline Ecological Risk Assessment Problem Formulation* (Anchor QEA 2014n).

Consistent with the results of these surveys, the receptors and exposure pathways (denoted in parenthesis in the following list) evaluated in the BERA were selected to represent organisms at different ecological trophic levels and with different site use and feeding strategies and consist of the following:

- Aquatic plants: phytoplankton (surface water)
- Invertebrates: zooplankton, benthic macroinvertebrates, and epibenthic invertebrates such as ribbed mussels and blue crab (surface water and surface sediment)
- Fish: striped bass and mummichog (surface water, surface sediment, and diet)[298]
- Semiaquatic birds: spotted sandpiper as representative of invertivorous birds, green heron and black-crowned night heron as representative of invertivorous/piscivorous birds, and double-crested cormorant and belted kingfisher as representative of piscivorous birds (surface water, surface sediment, and diet)
- Semiaquatic mammals: raccoon (surface water, surface sediment, and diet)

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[298] Atlantic menhaden are included as a prey item to assess risks to striped bass and aquatic-dependent wildlife.

In addition to quantifying risks to these receptors, the BERA also includes the following qualitative evaluations:

- Observations of fish and crab presence/absence, richness, and diversity
- Observations of bird and raccoon presence/absence, and for birds, richness and abundance
- Observations of reptile and amphibian presence/absence
- Observations of aquatic macrophyte presence/absence

The relationship between the exposure media and the receptors is presented in Figure 7-3. Given the number of LOEs evaluated in the BERA, overall risks were evaluated using a structured weight-of-evidence (WOE) approach described by USEPA (2016b). The approach identifies those LOEs that are more relevant, stronger, and more reliable than others and have a greater weight in an overall assessment of potential risk. In addition, the WOE evaluation determines whether the LOEs indicate evidence of risk, and (if so) what the magnitude of that risk might be. The outcome of the BERA WOE is presented as part of the conclusions in Section 7.2.6.

Bioaccumulation is the process by which chemicals accumulate in biological tissues, and biomagnification is the process by which chemicals increase with each trophic level, potentially reaching higher concentrations on a volumetric or mass basis compared with sediment and water. Bioaccumulation represents the final step of the process by which contaminants move from their sources to endpoints that are of primary importance for decision-making. In this case, the endpoints are the upper trophic levels of the food web and human consumption of fish and crab. Striped bass and blue crab are the primary species consumed by recreational anglers and crabbers, and the Atlantic menhaden, mummichog, and benthic invertebrates represent components of their food web. TPCB in striped bass and TPCB and dioxins/furans in blue crab are the primary CERCLA hazardous substances driving potential human health risk. Moreover, both chemicals are bioaccumulative. Because TPCB is the primary risk driver in both species, TPCB is the primary focus of the evaluation of bioaccumulation and biomagnification throughout the Study Area food web (see Section 4.10

for a discussion of tissue concentrations of TPAH [17] and Cu).[299] The following discussion outlines the approach to evaluating bioaccumulation and biomagnification in the Study Area.

Polychaetes and mummichog represent resident aquatic species in the Study Area and are components of the aquatic food web that includes striped bass and blue crab. Because these are resident species and they receive exposure primarily from surface sediment, TPCB concentrations in the tissue of these species are positively correlated with concentrations in local surface sediment.

In Newtown Creek, striped bass, blue crab, Atlantic menhaden, and white perch exhibit wide-ranging movement. Striped bass and blue crab are the primary species consumed by people. Atlantic menhaden and white perch were also sampled, because they represent lower-trophic-level migratory species and are important components of the food web. Striped bass move extensively throughout the New York Harbor region, in addition to annual migrations that include both time spent in the coastal ocean and upriver freshwater locations for spawning. While blue crab, including those of spawning age, may be present in Newtown Creek from spring to fall, some may leave to spawn during this time period. Newtown Creek is, however, within the range of salinities that both male and female blue crab inhabit at other times of the year. Overall, the soft-sediment bottom of Newtown Creek and its tributaries offer potential habitat for juvenile and mature female and male blue crab (Hines 2003; Rakocinski et al. 2003). As with striped bass, adult female blue crab collected in Newtown Creek likely reflect exposure that may have occurred over a broader geographical area than represented by the boundaries of the Study Area, while young male and female blue crab and adult male blue crab may have exposures that occurred over a smaller geographical region. Atlantic menhaden make extensive coast-wide migrations based on age and season (see Attachment F of Appendix I). They are likely to use the Study Area for only a portion of the year as non-breeding habitat.

Given these life history characteristics, TPCB concentrations in the tissue of these larger mobile and migratory aquatic species do not exhibit a straightforward correlation with

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[299] As noted in Section 6.6, for the evaluation of remedial alternatives in the FS, empirical data on surface sediment and/or tissue will be used to develop an approach for evaluating the extent to which each alternative may reduce risk from dioxins/furans, in addition to TPCB.

surface sediment concentrations (see Section 6.6). This pattern is consistent with exposure over a wide area, rather than exposure solely to local sediments collected in the vicinity of the fish sampling location.

The reference area tissue data provide one measure of the regional contribution to body burdens measured in the Study Area. TPCB concentrations measured in fish and crab collected from the Study Area exceed concentrations observed in the four Phase 2 reference areas. However, it should be noted that tissue concentrations are only available for the four Phase 2 reference areas, which were selected by USEPA from the 14 Phase 1 reference areas. Thus, concentrations measured in tissue in these areas may not be representative of the NY/NJ Harbor urban estuary as a whole. This conclusion is supported by the surface water data—Study Area surface water concentrations exceed those of the Phase 2 reference areas but are similar to East River data, indicating that the TPCB concentrations to which the mobile species collected in the Study Area are exposed may exceed those measured in the four Phase 2 reference areas. Thus, the four Phase 2 reference areas may provide an overly conservative (i.e., biased low) representation of the regional contribution to tissue concentrations in mobile fish and crab collected in the Study Area.

At this stage in the project, the contributions of exposures to elevated concentrations within the Study Area and regional conditions to the body burdens of migratory species cannot be estimated. The contributions of Study Area and regional sources to TPCB in fish and crab collected in Newtown Creek is an important consideration for remedial decision-making.

8.8 Summary

The CSM presented here summarizes the current state of knowledge regarding the environmental system of Newtown Creek and the physical, chemical, and biological processes that determine the transport of contaminants from sources to receptors. The CSM provides an integration of the previous discussions of nature and extent, sources, fate and transport processes, and risk assessment. The mass balance quantifies and integrates source magnitudes, fate and transport processes, and contaminant inventories. Differences among reaches can support the need for remedy selection and design on reach or sub-reach scales. As demonstrated in this report, the RI and FS Part 1 data were used to develop this CSM, which will be updated as the RI/FS progresses. Section 9 provides the overall conclusions of the RI.

9 CONCLUSIONS

This RI is a comprehensive study that sets a foundation to inform risk management decisions and evaluate remedial alternatives for the Newtown Creek Study Area. The RI complies with the AOC entered into with USEPA for this site.

The very comprehensive dataset collected within the Study Area and from 14 reference areas during Phase 1 and Phase 2 of the RI, as well as Part 1 of the FS field program, supports the development of a CSM that describes the following:

- The nature and extent of contamination in various environmental media in the Study Area[300]
- The degree to which these media are affected by ongoing point and non-point sources and by contaminant fate and transport processes that affect the spatial and temporal distribution of contamination
- The resultant risks to human health and ecological receptors from exposure to CERCLA hazardous substances and other stressors in the Study Area, which are consistent with an urban industrialized waterway

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[300] As described in Sections 4.1.2, 4.12, and 8.4, this RI Report CSM focuses on three chemicals: TPAH (17), TPCB, and Cu, for a number of reasons. TPAH (17) is a primary risk driver in the BERA (see Appendix I), TPCB is a primary risk driver in the BHHRA and BERA (see Appendices H and I, respectively), and Cu was selected as a representative metal, because the BERA estimated HQs greater than 1 (but less than 3) for some receptors. TPAH (34), C19-C36, 2,3,7,8-TCDD, and Pb are included in the nature and extent evaluation for all media in which samples were analyzed for these chemicals in Section 4, because TPAH (34) and C19-C36 are important contributors to benthic toxicity (as determined by USEPA subsequent to completion of the BERA), 2,3,7,8-TCDD was identified as a risk driver in the BHHRA and as a COPEC in the BERA, and Pb was identified as a COPEC in the BERA. Although dieldrin was not identified as a COPC or a COPEC, it was included in the nature and extent evaluation of surface sediment and tissue because of elevated concentrations in polychaete tissue in one reach of the Study Area (English Kills). However, these five contaminants (i.e., TPAH (34), C19-C36, 2,3,7,8-TCDD, Pb and dieldrin) were not included for evaluations of sources and fate and transport in Sections 5 and 6, respectively, because: 1) their distributions in environmental media (including surface sediment) are similar to those within the same class (i.e., hydrocarbons, bioaccumulative organics, and metals); 2) in some of the media, some of the contaminants (e.g., C19-C36, 2,3,7,8-TCDD, and Pb) were infrequently detected; and 3) their fate and transport characteristics (i.e., partitioning behavior) are similar, especially to others in the same class, such that they can be represented by TPAH (17), TPCB, and Cu.

Sections 9.1 through 9.3 summarize the important findings of this RI, first on a Study Area-wide basis and then on a reach-specific basis, because differences among the various segments of the Study Area will be important when evaluating remedial alternatives during the FS.

9.1 Study Area-Wide Summary

Physical characteristics. The Study Area can be described as a main channel that extends inland from the East River approximately 2.8 miles to the juncture of the East Branch and English Kills tributaries. Coupling the main stem and the English Kills tributary, the total length of the system is 3.8 miles. The main stem is joined by five tributaries along its length. The arithmetic average depth of the main channel at mean tide level is 18 feet in CM 0 – 2 and 16 feet in CM 2+. Depths are shallower along the shoreline as compared to the central portion of the main channel, consistent with the bounds of the navigation channel. The Study Area is tidally influenced, with semidiurnal changes in surface water elevation of approximately 4 to 6 feet.

Although there are differences among the five tributaries (described in Sections 4, 6, 7, and 8), all five tend to be narrower than the main channel and shallower in depth, due primarily to the preferential deposition of solids that enter these tributaries from point sources. Each of the five tributaries can be characterized by the presence of a major CSO (or WWTP treated effluent overflow in the case of Whale Creek) at the head of the tributary and a lack of any natural fresh surface water inflow. The influence of the East River, including solids loadings carried into the Study Area due to tidal exchange, varies in these tributaries, with those more proximal to the East River (Whale Creek and Dutch Kills) being influenced greater than those farther upstream (Maspeth Creek, English Kills, and East Branch).

The land use around Newtown Creek from the 1800s through the present has been predominately industrial. This industrial development occurred in parallel with municipal use of Newtown Creek as a receiving waterbody of both stormwater and wastewater discharges. The municipal use of Newtown Creek as a receiving waterbody for stormwater, as well as sanitary and industrial wastewater discharges has evolved over time, especially with the initial construction of a WWTP in the late 1960s. Newtown Creek continues to be a major receiving waterbody of these types of discharges, as well as CSO discharges (containing combined flows of stormwater and sanitary and industrial wastewater), and treated effluent from the Newtown Creek WWTP overflow during rainfall events. Newtown Creek is also a designated SMIA, which will continue to give preference to industrial uses in upland areas. Consequently, the land use history and urban landscape in which Newtown Creek exists shapes the CSM and informs the nature and extent of COPCs and potentially significant sources, as well as key fate and transport characteristics, pathways, and exposure scenarios.

Reference areas. The nature and extent of contaminants in Newtown Creek surface water and sediment, including TPAH (17), TPCB, and Cu, should be viewed in the context of its urban industrial setting. This setting is characterized in this RI using samples collected from 14 Phase 1 and 4 Phase 2 reference areas, all of which are tidal inlets in the New York Harbor area. The Study Area will continue to be subject to ongoing influences from human activities that are specific to this urban environment in the form of ongoing discharges of solids from CSO, MS4, and other stormwater outfalls, particularly in the tributaries.

Sources. The nature and extent of contaminants in Study Area sediment are the result of historical discharges, current point and non-point sources, groundwater-related input, East River tidal exchanges, and changes in chemical loading over time.

Chemicals are currently entering the Study Area from multiple potential external sources, including the following:

- Point sources and overland flow
- East River
- Groundwater
- Other minor sources
    - Bank erosion
    - Atmospheric deposition
    - Overwater activities
    - Contaminant seeps

The most significant ongoing sources of contaminants to surface sediment in the Study Area, including TPAH (17), TPCB, and Cu, are from point sources (primarily CSOs, stormwater, and the Con Edison – 11th Street Conduit outfall), the East River, and groundwater flow (due to vertical discharge through the sediment; lateral discharge to the surface water will be further evaluated as part of the FS). Newly depositing solids from current ongoing sources mix with solids in the surface sediment bed, resulting in surface sediment contaminant concentrations that reflect a blend of those from new sediment and previously deposited sediment; the extent of this blending varies with location in the Study Area.

Fate and transport processes. Hydrodynamic processes (i.e., tidal currents and density-driven circulation) generate relatively low near-bed current velocities throughout large portions of the Study Area. The low energy environment creates a net depositional environment throughout the Study Area. Deposition has been ongoing for decades. Historically, contaminant loads to the surface sediments were likely much greater, as evidenced by the higher contaminant concentrations in subsurface sediment at many locations. The current distribution of contaminants in the sediment column of the Study Area is due to uncontrolled historical and ongoing sources, historical fate and transport processes, and changes in contaminant loads over time; therefore, the locations of sediment impacts observed today cannot necessarily be directly linked to proximate upland sites or point sources.

The low energy environment also results in minimal or no erosion of the sediment bed. Episodic erosion in specific areas or reworking of the upper layer of the bed may occur in some locations. In the tributaries, high velocities near CSO discharges during high-flow storm events can resuspend sediment, which can then be transported downstream of the discharge. In the main stem, vessel propwash associated with shipping activities can also act to resuspend sediment, although based on modeling efforts completed to-date, a majority of that sediment redeposits in the same general area.

Because many contaminants, including TPAH (17), TPCB, and Cu, sorb strongly to particles in the water, their transport and dispersal depends in part on the settling and resuspension properties of the particulate matter. Larger particles settle closer to the release point, and finer particles are generally transported farther. In general, contaminants sorb more strongly to finer particles, compared with coarser particles. This process contributes to the observed distribution of contaminants in surface sediment of the tributaries—the generally observed increase in contaminant concentrations with distance downstream from the head of the tributaries is likely due (in part) to differential settling of fine and coarse contaminated particulate matter.

A key finding of the RI is that contaminant concentrations in the surface sediment layer have likely been declining at many locations over time. The mechanism is the mixing of newly deposited solids with previously deposited solids. This process is occurring, and has occurred in the past, in the surface sediment throughout the Study Area, resulting in blended surface sediment concentrations. Based on the vertical distribution of contaminants in sediment, including TPAH (17), TPCB, and Cu, it is also apparent that chemical loads to the sediment have decreased over time, resulting in lower concentrations in surface sediment compared with subsurface sediment. The rate of decline in surface sediment chemical concentrations, as inferred from differences in surface and subsurface sediment concentration data, varies by reach, based on the source of the newly depositing solids, the rate of sediment mixing, the rate of sedimentation, and the concentration of the surface sediment at that point in time when it and the newly deposited sediment are mixing.

Although localized physical disturbances (e.g., propwash) cause temporary resuspension of sediment, and newly deposited solids mix with previously deposited solids in the surface sediment, the subsurface sediment appears relatively stable. Potential future navigational dredging represents an additional process that could affect bed stability in project areas, depending on how much and the manner in which such dredging would be performed.

Groundwater. In addition to solids and contaminant loadings from ongoing point sources and the East River directly depositing on the surface sediment in the Study Area, groundwater is a potential ongoing source of chemicals to subsurface sediment via transport through the interstitial spaces (as porewater) and sorption onto the solid matrix. Groundwater discharge to the Study Area consists of discharge to the base of the Study Area and lateral groundwater flow through vertical permeable shorelines[301] to the surface water (i.e., lateral discharge). Groundwater discharge to the base of the Study Area provides

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[301] Vertical permeable shorelines include pile-support concrete, precast concrete blocks, and vertical wood bulkheads.

chemical loads to subsurface sediment. The estimated annual loads of TPAH (17), TPCB, and Cu from groundwater to Study Area subsurface sediment were calculated as the product of groundwater discharge (i.e., flow rate) and dissolved phase groundwater chemical concentration measured within the native material. This evaluation showed that there is some chemical load from groundwater to the Study Area, but it varies spatially by reach. The chemical loads are associated with groundwater entering the Study Area at the base and edge of the subsurface sediment bed. Due to attenuation within the subsurface sediment, however, the chemical loads associated with groundwater entering the base of the Study Area (i.e., at the bottom of subsurface sediment) appear to contribute negligible chemical loads to the surface sediment (which includes the biologically active zone); this process will also be further evaluated using the CFT model. The effects of contaminant loads from groundwater discharge through vertical permeable shorelines to surface water were undiscernible based on the evaluation of dry weather surface water chemistry; however, because shallow lateral groundwater discharge inputs to Newtown Creek have not been empirically characterized, the process will be further evaluated in the FS through data collection planned by USEPA and through sensitivity analyses with the CFT model.

NAPL. For much of the Study Area, where NAPL was observed, it was primarily in a residual state (i.e., shake test bleb results, visual observations of NAPL blebs in sediment and/or native material), and was distributed intermittently in sediment (i.e., located sporadically throughout an area, not clustered at a particular location). Sheen was observed intermittently in sediment throughout the Study Area. More significant Category 2/3 NAPL observations (i.e., shake test layer results, visual observations of NAPL coated and NAPL saturated sediment and/or native material) were limited to cores collected in three discrete areas: CM 1.6 to 1.7 (main stem), CM 2.4 to 2.7 (Turning Basin), and CM 2.9 to 3.2 (lower English Kills).

NAPL mobility testing of CM 0 – 2 subsurface sediment and native material performed during the FS Part 1 field program showed that NAPL in CM 0 – 2, where present, is immobile. Quantitative NAPL mobility testing for CM 2+ and the tributaries was performed as part of the FS Part 2 field program (data for the FS Part 2 field program are not included in the RI Report, but are presented in the FS NAPL DER [Anchor QEA 2022a]).

Ebullition. Surveys of gas ebullition and ebullition-generated sheens were conducted in August 2015 and September 2016, as part of the Phase 2 investigations. Both studies made observations of the location, frequency, and magnitude of bubble generation and sheen blossoms at the water surface, to develop an understanding of conditions where gas ebullition-facilitated NAPL transport is most likely to occur. These FESs were performed during the time of year when gas ebullition was expected to be most active (i.e., in summer when the surface water temperature is warmest and during very low tide conditions), and the surveys are considered a conservative record of observations compared to other times of the year. A gas ebullition-facilitated NAPL/contaminant transport pilot study was conducted in September 2017 to develop and test methodologies for the July and October 2018 and January 2019 gas ebullition field programs.

Additional investigation of gas ebullition-facilitated transport of NAPL/contaminants was initiated in July 2018 and October 2018 and continued with a visual survey event in January 2019 and ongoing camera observations, as part of the FS gas ebullition field program. The program includes additional quantitative NAPL/contaminant and gas flux measurements (July and October 2018), visual observations of sheens and gas bubbles (January 2019 visual survey and ongoing camera observations), sediment temperature depth profile measurements, and additional work during different times of the year to capture the effect of different temperatures on gas ebullition-facilitated transport of NAPL/contaminants. Following this program, the results will be used to extrapolate flux measurements to other times of the year and/or other areas of the Study Area for use in CFT modeling, refinement of the RI/FS CSM, and potentially to support the development and technology screening of FS alternatives. The results of the 2018 to 2019 FS field program are provided in the FS Gas Ebullition DER (Anchor QEA 2022b).

Risk. Humans and ecological receptors are exposed to contaminated media in the Study Area through a number of exposure pathways. Baseline human health and ecological risk assessments were completed within the Study Area and included evaluation of risks in the four Phase 2 reference areas. Human health risks considered included a variety of current and future scenarios by which people may be exposed to environmental media within Newtown Creek, including consumption of fish and crabs caught by recreational anglers/crabbers. Ecological risks were evaluated for a number of representative receptor categories, including aquatic organisms that inhabit the surface sediment (i.e., benthic invertebrates), organisms exposed to the surface water, and higher trophic organisms (i.e., organisms that occupy higher levels in the food chain, such as crab, fish, birds, and mammals that come into contact with the creek).

The BHHRA evaluated 12 exposure scenarios; of these, risks in excess of USEPA's acceptable cancer risk range and non-cancer hazard threshold were identified for only the following exposure scenarios:

- Consumption by recreational anglers/crabbers of fish and crab tissue obtained from the Study Area, primarily due to tissue concentrations of PCBs in fish and PCBs and dioxins/furans in crab.
- General construction worker exposure to surface sediment along the shoreline in certain areas of the Study Area, where the estimated potential for noncancer hazards is above the HI threshold of 1. This results from the sum of the risks from all COPCs (i.e., no individual COPCs exceeded the HI of threshold of 1). PCBs in surface sediment in these localized areas are the largest contributor to the HI exceedance.
- Risk characterization for reference area fish and crab consumption determined that the estimated reference area cancer risks are at the upper end of USEPA's acceptable risk range, or exceed the acceptable risk range, and noncancer HIs exceed the threshold of 1.

A quantitative and qualitative WOE approach was used to evaluate multiple LOEs in the BERA. This comprehensive approach identified risks that were, for the most part, limited to CM 2+ and the tributaries. These risks are associated with the following receptors and exposure pathways:

- Surface sediment toxicity to benthic organisms in CM 2+ and the tributaries is greater than toxicity in sediment in the four Phase 2 reference areas, based on the reference area toxicity threshold. Although toxicity in many of these sample locations may be associated with PAHs in porewater and in bulk sediment, with some contribution from porewater metals (Cu, Pb, and Zn), there is a subset of samples for which toxicity cannot be attributed to either PAHs or metals in porewater. At these locations, toxicity test results appear to be confounded by stressors common to an urban setting, such as elevated levels of porewater sulfide and/or elevated levels of complex hydrocarbon mixtures in sediment (C19-C36). Benthic toxicity in CM 0 – 2 is similar to that measured in the four Phase 2 reference areas based on the 28-day toxicity tests, but greater than reference area results for the 10-day toxicity tests.

- For benthic organisms, due to TPAH (34) and C19-C36 in bulk sediment, as well as PAHs, Cu, Pb, Zn, and TPCB congeners in porewater; for spotted sandpiper, due to Cu, Pb, and TPCB congeners in its diet; and for green heron, black-crowned night heron, and belted kingfisher, due to TPCB congeners in their diets.
- Using a tissue residue approach and NCG CBRs, all COPEC HQs were less than a threshold of 1. When USEPA Region 2 CBRs were used, HQs greater than a threshold of 1 were exceeded for bivalves and polychaetes, due to PAHs and TPCB congeners; for blue crab, due to TPCB congeners and Cu; for striped bass, due to TPCB congeners, dioxin/furan TEQ, and 2,3,7,8-TCDD; and for mummichog, due to TPCB congeners and Cu.

Migratory fish and crab species that visit the Study Area receive a portion of their body burden from regional sources of contamination, rather than solely from sources specific to Newtown Creek. In general, PCB concentrations in reference area sediment and biota tissue are lower than in Study Area sediment and biota tissue, indicating a site-specific component to body burdens. Specifically, the cancer risks and noncancer hazards calculated for the Phase 2 reference areas provide one estimate of regional risks that could be present in the absence of Study Area-related contamination. The component of the body burden of these species that would continue, regardless of any reductions in exposure achieved within the Study Area, is an important consideration for remedial decision-making.

In addition to the risks quantified as part of the baseline risk assessments, additional risks from confounding or other factors mentioned previously can also pose risks to human health and the environment. For example, the BHHRA focuses solely on standard CERCLA hazardous substances and may underestimate total human health risks. Similarly, there are confounding factors that appear to influence toxicity to the benthic invertebrates in the tributaries at a subset of locations. These additional stressors should be considered, along with the results of the baseline risk assessments, when evaluating achievable risk reductions.

9.2 Reach-Specific Summary

A key finding of this RI is that the reaches of the Study Area (CM 0 – 2, CM 2+, and each tributary) differ materially in physical characteristics, contaminant distributions, sources of solids and contaminants, relative contributions of historical versus ongoing sources, fate and transport processes, and risk. Those differences will play an important role, and will pose unique challenges, when developing and assessing remedial alternatives in the FS. The following subsections summarize key conclusions that can be drawn from the RI information related to reach-specific characteristics and the comparison of conditions in these reaches with conditions in reference areas, sources, fate and transport, and risks.

Incremental mass loading from groundwater to sediment is negligible in comparison to the mass of chemicals already present in sediment. In addition, groundwater chemical loads attenuate substantially within the subsurface sediments; therefore, porewater loads from subsurface sediment to surface sediment are relatively low. As summarized in Section 8.5.2.3, chemical loading from groundwater varies across the Study Area, but the contribution of groundwater loads to surface sediment and surface water are negligible and are not expected to be an important factor in remedy evaluation and selection. The contribution of chemical loads from groundwater will be further evaluated with the CFT model.

The nature and extent of contamination in shallow surface sediment within the Study Area is affected by influences including the following: ongoing deposition of solids from point sources; surface water and solids exchange with the East River due to the tides; mixing (due to biological activity within the surface sediment [i.e., bioturbation])[302]; episodic storm events in the tributaries near the large outfalls; and marine vessel traffic, which also acts as a sediment mixing process. These influences contribute to the following notable findings regarding the distribution of contamination[303]:

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[302] See Section 6.4.4.4 for details on physical mixing between surface sediment and surface water, and the degree to which the surface layer of the sediment bed in the Study Area may be mixed due to bioturbation and physical processes.

[303] As described in Section 4, this report focuses on TPAH (17), TPCB, and Cu, although the risk assessments (see Section 7 and Appendices H and I) and the evaluation of nature and extent (see Section 4) provide results for all analytes that are CERCLA hazardous substances.

- **CM 0 – 2**
  - Concentrations of contaminants, including TPAH (17), 2,3,7,8-TCDD, TPCB, and Cu, in surface sediment in CM 0 – 1 are generally the lowest in the Study Area, are primarily influenced by solids transported into the Study Area by the East River, and are consistent with the reference area dataset from both Phase 1 and Phase 2 sampling programs. Concentrations in CM 1 – 2 are higher than those in CM 0 – 1, but are also consistent with (or approaching) the reference area data.
  - Concentrations of contaminants, including TPAH (17), TPCB, and Cu, tend to increase with depth in the subsurface sediment, below the surface sediment.
  - Deposition of solids in this downstream portion of the Study Area is primarily from East River tidal exchange, with some minor contribution from point sources in this reach. Solids that enter the CM 0 – 2 reach, primarily from East River tidal exchange, become mixed within the shallow surface sediment layer via biological and physical processes, including vessel propwash. Due to the net depositional nature of this reach, surface sediment concentrations are more similar to the East River than surface sediment concentrations found in CM 2+ and the tributaries. Regardless of remediation efforts targeted at decreasing existing surface sediment concentrations, the East River source in CM 0 – 2 is sufficiently dominant such that surface sediment concentrations in this reach are likely to be similar to the East River and reference waterbodies influenced by similar CSO, municipal, and industrial stormwater discharges.
  - NAPL was not observed in surface sediment, and where present in the subsurface sediment, NAPL was in a residual state and distributed intermittently. Sheen was observed intermittently in surface and subsurface sediment. At CM 1.7, thin discontinuous lenses of NAPL were reported in the subsurface in a limited number of cores. NAPL mobility testing completed throughout CM 0 – 2 (including CM 1.7) during the FS Part 1 field program concluded that NAPL in both subsurface sediment and native material in CM 0 – 2 is immobile.
  - Minimal gas ebullition and concomitant sheen blossom formation was observed in CM 0 – 2 during field surveys conducted in 2015 and 2016 to characterize the presence and extent of ebullition-facilitated NAPL transport to the surface water.
  - Toxicity to benthic macroinvertebrates based on a 28-day toxicity test and risks to other ecological receptors such as fish and crab are generally similar to the four

  Phase 2 reference areas. However, there are differences. For example, test organism survival in 28-day toxicity tests was above the reference envelope threshold in CM 0 – 2, but was below the threshold in 10-day toxicity tests throughout much of the creek, including CM 0 – 2.
  - Although there is some evidence of harm in CM 0 – 2 based on some LOEs, the overall WOE evaluation indicates that the magnitude of response is lower in this segment of the Study Area relative to CM 2+ and the tributaries.
  - The range of surface water concentrations of TPAH (17), TPCB, and Cu in CM 0 – 2 overlaps with the range of concentrations measured outside the Study Area in the East River. East River surface water samples were collected from a location approximately 0.2 mile west of the mouth of Newtown Creek, and from three locations along a transect at the mouth of Newtown Creek (see Figure 5-29).

- **CM 2+**
  - Concentrations of TPAH (17), TPCB, 2,3,7,8-TCDD, and Cu in surface sediment are higher than in CM 0 – 2 and above reference area concentrations.
  - Concentrations of contaminants, including TPAH (17), TPCB, and Cu, tend to increase with depth in subsurface sediment below the surface sediment.
  - Solids deposited from CSOs, stormwater, and East River tidal exchange become mixed with previously deposited solids in the surface sediment layer via biological and physical processes. The contaminants associated with these newly depositing solids mix with the contaminants associated with previously deposited solids, resulting in a blend of currently and previously deposited contaminants in the surface sediment.
  - NAPL was observed in CM 2+, with a greater magnitude of NAPL reported along the western perimeter and in the southwest corner of the Turning Basin in subsurface sediment and native material, and less frequently in surface sediment. Sheen was observed in surface and subsurface sediment at a number of locations in this reach.
  - During the 2015 and 2016 FESs and the 2017 pilot study, areas of gas ebullition and sheen blossom formation were observed in the Turning Basin along the Brooklyn and Queens shorelines, at water depths shallower than six meters. The occurrence of gas bubbles and sheen blossoms generally increased with lower tidal

  elevations, with the maximum gas ebullition rate observed within a 1-hour window around low tide.
  - Toxicity to benthic macroinvertebrates and risks to other ecological receptors such as fish and crab are greater than in the Phase 2 reference areas. Toxicity to benthic macroinvertebrates at some locations cannot be attributed solely to porewater concentrations of PAHs, PCBs, and metals, but may also be influenced by other stressors such as elevated porewater sulfide and/or elevated levels of complex hydrocarbon mixtures (C19-C36).

- **Tributaries**
  - There is no natural fresh surface water inflow to the tributaries; most fresh surface water flow is due to municipal and industrial point sources and overland flow. Tidal exchange with East River water occurs in the tributaries, with greater influence from the East River occurring in the lower tributaries (Whale Creek and Dutch Kills) than the upper tributaries (Maspeth Creek, English Kills, and East Branch).
  - Major CSOs are present at the heads of English Kills, East Branch, Maspeth Creek, and Dutch Kills (the Newtown Creek WWTP treated effluent overflow outfall discharges to Whale Creek) and are the primary ongoing source of solids to the tributaries, although solids originating from the East River also contribute to deposition in the tributaries (to a large extent in Dutch Kills and Whale Creek).
  - Surface sediment exhibits higher TOC levels than normally found in natural systems, due primarily to discharges of solids from CSO and MS4 point sources.
  - Concentrations of TPAH (17), TPCB, 2,3,7,8-TCDD, and Cu in surface sediment are generally higher than in CM 0 – 2 and are above reference area concentrations. Concentrations differ among the individual tributaries in many cases.
  - Concentrations of some chemicals in surface sediment decline toward the heads of the tributaries, likely due to mixing of solids and contaminants from upstream and downstream sources and differences in settling rate between fine- and coarse-grained solids.
  - Concentrations of contaminants, including TPAH (17), TPCB, and Cu, tend to increase with depth in subsurface sediment below the surface sediment.
  - NAPL was not observed in Whale Creek or Dutch Kills and only reported in a few cores in Maspeth Creek, East Branch, and upper English Kills. Varying amounts

of sheen were observed in surface and subsurface sediment in each of the tributaries. In a localized area within lower English Kills, a greater magnitude of NAPL was observed, primarily in lenses of coarse-grained material in the subsurface sediment and native material.

- During the 2015 and 2016 FESs and the 2017 pilot study, areas of gas ebullition and sheen blossom formation were observed in each of the tributaries. More widespread gas bubbles were observed in the tributaries, where the TOC is higher and water depths are generally shallower than the deeper water in the main stem.
- Toxicity to benthic macroinvertebrates and risks to other ecological receptors such as fish and crab are greater than in the Phase 2 reference areas. Toxicity to benthic macroinvertebrates at some locations cannot be attributed solely to porewater concentrations of PAHs, PCBs, and metals, but may also be influenced by other stressors such as elevated porewater sulfide and/or elevated levels of complex hydrocarbon mixtures (C19-C36).
- DO concentrations are generally lower in the tributaries than in the main stem, particularly during the summer months, and are a factor contributing to a benthic community that is stressed relative to conditions in the main stem of the creek and the Phase 2 reference areas.

9.3 Summary

Surface sediment contamination influences the ecological and human health risks within the Study Area. The RI data demonstrate that surface sediment contamination varies in composition and concentration by reach in the Study Area. This variation is primarily due to the influence of the East River and other ongoing sources to the Study Area. These regional and site-specific ongoing external inputs to Newtown Creek will continue to reflect contributions from ongoing urban sources to the Study Area that include, but are not limited to, tidal flows from the East River, point source discharges, overland stormwater flow, and other contaminant sources (including shoreline seeps, lateral groundwater discharge, atmospheric deposition, shoreline erosion, and overwater activities) that contribute contaminant loads to the Study Area to varying degrees and often vary locally and occur intermittently. These ongoing contaminant contributions to the Study Area will place practical limits on the feasible future reductions of contaminant concentrations in surface sediment, surface water, and tissue; will influence baseline ecological and human health risks in the Study Area; and may disrupt targeted remedy outcomes in some portions of the Study Area, such as tributaries. Consideration should be given in the FS to remedial approaches, timing, and long-term effectiveness, as the East River, CSO and stormwater discharges, other point sources, and overland stormwater flows will continue to contribute a significant ongoing load of constituents (e.g., TPAH [17], TPCB, and Cu) to the Study Area. In addition, some upland properties may potentially contribute constituents to the Study Area. The FS will need to evaluate these ongoing contributions consistent with USEPA's first-listed risk management principle, which states that significant direct and indirect ongoing sources should be identified and controlled if they have the potential to cause recontamination at sediment sites (Horinko 2002). As noted by USEPA guidance, "Identifying and controlling contaminant sources typically is critical to the effectiveness of any Superfund sediment cleanup (USEPA 2005a). Remedial consideration also will necessarily reflect that the confines of the Study Area are coterminous with the CWA/LTCP planning area, and the parties, including the agencies, must balance the remedial decisions under both statutes.

Notable differences among the reaches of the Study Area in surface sediment contaminant concentrations, human and ecological exposure, contaminant sources, and fate and transport must play an important role when developing remedial alternatives. Consistent with USEPA guidance (USEPA 2005a), at large, complex sites like Newtown Creek, alternatives that combine a variety of approaches and take into consideration site-specific characteristics should be considered. As such, when assembling alternatives, the FS should consider a combination of remedial technologies, including the following:

- Monitored natural recovery (MNR) in areas where ongoing recovery is occurring and where remedial goals can be accomplished through natural recovery processes over time
- Enhanced natural recovery (ENR) to accelerate the process in areas where natural recovery is occurring
- Targeted dredging, capping, and in situ stabilization in areas where MNR and ENR are determined to be ineffective or impractical approaches
- Consideration of NYC's LTCP and ongoing CSO and MS4 compliance under CWA, as well as how ongoing releases from these systems will affect the sediment remedy in terms of achieving risk reduction

The RI represents a comprehensive study that complies with the AOC entered into with USEPA for this site. The voluminous dataset supports multiple LOEs to characterize the nature and extent of contamination in the Study Area. This work also establishes a solid foundation to evaluate a combination of sustainable remedial approaches to utilize in different portions of the creek to achieve practicable risk reduction and ensure long-term success. For the purposes of accomplishing the RI objectives, there are no further data limitations or gaps that need to be filled for RI completeness. The results of Phase 1 and Phase 2 of the RI pointed to additional data needed to support the FS, such as further tissue sampling; NAPL delineation refinement and mobility testing; and additional, more quantitative gas ebullition studies. The results of Part 1 of the FS field investigation are included in this RI Report and the RI CSM. The results of Part 2 of the FS field investigation will be important for refining the CSM, and together with the results of Part 1 of the FS field investigation, will be used to develop a comprehensive suite of remedial alternatives and evaluate remedy effectiveness.

The RI meets the objective of identifying potentially significant sources, but localized sources may still exist. There may be some upland data gaps, due to lack of information about upland sites; however, it is anticipated that any key sites will be identified by state regulatory agencies, and the necessary data will be collected under the appropriate regulatory program to close this data gap. Thus, the RI Report does not draw conclusions as to whether all potential upland sources of COPCs/COPECs to Newtown Creek have been identified. For example, elevated concentrations of PCBs in surface sediment have been observed in discrete locations within the Newtown Creek system, particularly within Dutch Kills, English Kills, and CM 2+ (see Table 4-16), but upland sources have not been identified. This does not represent an RI data gap, because upland sites are outside the Study Area, but upland sites may represent an ongoing source to the Study Area. As additional information becomes available, it may be incorporated into the FS. Notwithstanding the extensive dataset compiled during this RI, future investigations undertaken within the boundaries of the Study Area may indicate as yet unidentified sources that will need to be considered as remedial designs move forward.

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GENERAL TABLE, FIGURE, AND CHEMISTRY DATA NOTES

General Table, Figure, and Chemistry Data Notes

The notes that follow contain information regarding chemistry data presented in the *Remedial Investigation Report* (RI Report). These notes are intended to supplement notes included within the text, figures, and tables throughout the RI Report. These notes are globally applicable to all figures and tables of the RI Report, including the appendices, unless otherwise specified.

- Units for solids and chemistry data
  - Chemistry results from all samples of **solids** (sediment, native material, sediment trap, particulate phase point source, and particulate phase surface water samples) are reported on a **dry weight** basis. Results reported as "percent" for solid samples are on a weight percent (wt%) basis.
  - Chemistry results from all samples of **tissue** are reported on a **wet weight** basis. Results reported as "percent" for tissue samples are also on a wt% basis.
  - Percent fines results for all **solids** samples are reported on a **dry weight** basis. Results reported as "percent" for these samples are on a wt% basis.
  - Gross solids deposition rates are in units of mg/cm^2/day, reported on a **dry weight** basis.
- Chemical-specific notes
  - Percent fines correspond to particle sizes of less than 62.5 micrometers (µm).
  - Measurements of total polychlorinated biphenyls (TPCB) presented in the RI Report vary according to media, as follows:
    - Surface sediment, subsurface sediment, and native material samples consist of a combination of measurements of TPCB congeners and TPCB Aroclors, the latter of which were adjusted to make the two types of measurements consistent (see Section 4.1.3.1 of the RI Report for more details).
    - Sediment trap samples were analyzed for TPCB congeners.
    - Surface water samples were analyzed for TPCB Aroclors (Phase 1 only) as well as TPCB congeners (Phase 1 and Phase 2). These data are discussed and presented separately (see Section 4.7.3.3 of the RI Report).
    - Porewater and groundwater samples were analyzed for TPCB congeners.
    - Point source samples were analyzed for TPCB congeners.

    - Tissue samples were analyzed for TPCB congeners.
    - Air samples were analyzed for TPCB Aroclors.
  - Data reported by the laboratories as benzo(k)fluoranthene include the co-eluting isomer benzo(j)fluoranthene. The chemical names shown in figures and tables were updated accordingly; the results are reported as benzo(j,k)fluoranthenes.
  - Phenol was analyzed using the U.S. Environmental Protection Agency Method 8270D (SW-846) Semivolatile Organic Compounds by Gas Chromatography/Mass Spectrometry.
  - C19-C36 aliphatic hydrocarbons are reported as an "unadjusted" result only because this is a fractionated aliphatic range that is integrated from n-nonadecane to n-hexatriacontane. Due to it being fractionated, the aromatics do not need to be subtracted (they are collected in a separate fraction), so no "adjusted" range is needed in this aliphatic fraction range. Note that this is consistent with the Massachusetts extractable petroleum hydrocarbon methods.[305]
  - Figures and tables of sediment total organic carbon (TOC) in the main text of the RI Report include Feasibility Study (FS) shoreline, National Grid, Phase 2, reanalyzed Phase 1, and original, corrected Phase 1 TOC data. Figures and tables in the appendices include only FS shoreline, National Grid, Phase 2, and reanalyzed Phase 1 TOC data. See Section 4.1.3.3 of the RI Report for more details.

- Chemical name acronym definitions:
  - BHC = benzene hexachloride
  - BTEX = benzene, toluene, ethylbenzene, and xylenes
  - cPAH = carcinogenic polycyclic aromatic hydrocarbon
  - DDD = dichlorodiphenyldichloroethane
  - DDE = dichlorodiphenyldichloroethylene
  - DDT = dichlorodiphenyltrichloroethane
  - HPAH = high molecular weight polycyclic aromatic hydrocarbon
  - LPAH = low molecular weight polycyclic aromatic hydrocarbon
  - PAH = polycyclic aromatic hydrocarbon
  - PCE = perchloroethylene
  - PCB = polychlorinated biphenyl

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[305] http://www.mass.gov/eea/docs/dep/cleanup/laws/ephmcp.pdf

  - SEM = simultaneously extracted metals
  - TEQ = toxic equivalent
  - TOC = total organic carbon
  - Total DDx = sum of the 6 DDx compounds: 2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT
  - Total PAH (17) = sum of 17 individual PAHs
  - Total PAH (34) = sum of 34 individual PAHs
  - TSS = total suspended solids

- Chemistry statistic tables
  - Statistics for chemical results presented in the RI Report and its appendices were calculated using detected and non-detected results. Non-detected results are set to the method detection limit (MDL).
  - For chemicals that have fewer than two detected results in a given reach, the minimum and maximum results are shown and the remaining statistics were not calculated.
  - In cases where all values are non-detect, the minimum and maximum values reported are set to the MDL; the values may differ due to different MDLs for different samples.

- Kaplan-Meier (KM) totals
  - KM totals were calculated using the *Nondetects and Data Analysis* package (Helsel 2005; Lopaka 2013) in the R statistical computing language (R Core Team 2015). The procedures for calculating totals using the KM method are as follows:
    - KM totals were calculated as the intermediate KM mean, multiplied by the number of constituents in the total.
    - If there were fewer than three detected constituents, the KM total was not calculated.
    - If the smallest value contributing to the total was a non-detect, the value was treated as detected (Efron’s bias correction), and the KM total was reported as estimated.
    - If the largest value contributing to the total was a non-detect value, the value was treated as detected, and the KM total was reported as a non-detect.

        - If the sum of the detected values (weighted as applicable) was less than the sum of the non-detect values (weighted as applicable), the KM total was reported as a non-detect.
        - Rejected values were not included in the KM total. If all constituents of the total were rejected values, the KM total was not calculated.
        - Where the KM total was not calculated, the non-detected results are not included in the total calculation.
    - See Section 3.2.4 and Attachment D6 of the BHHRA (see Appendix H) for more information on KM totals.

- Description of presentation tools
    - The five primary types of graphics used throughout the RI Report and its appendices are briefly described as follows:
        - **Plan view maps** show data at individual sampling locations. Their primary purpose is to show the spatial patterns of the data. Therefore, quantiles of the data are used to facilitate visual observation of spatial patterns. Data values at each location are presented as color-coded symbols; each color represents a specific range of concentrations, based on quantiles of the data (e.g., 20th, 40th, 60th, and 80th percentiles). In most cases quintiles are used; however, quartiles are sometimes used when there is a small number of samples or unique values. In some cases, these quantiles were modified to provide rounded values. For presentation of subsurface sediment data, plan view maps showing stacked bars color coded by concentration ranges are used to display concentration by depth at each core location.
        - **Longitudinal profiles** show individual chemical concentrations throughout the Study Area. Their primary purpose is to show the spatial patterns of the data as a function of distance from the confluence of Newtown Creek with the East River. For comparison, the concentrations from the Phase 1 and Phase 2 reference areas are shown in these figures, in most cases. Concentrations are presented on the vertical axis, versus the creek mile (CM) on the horizontal axis; CM 0.0 is the mouth of the creek at the confluence with the East River. Therefore, a sample in a tributary and a sample in the main stem may have the same CM, even though they are separated in space. The main stem and the

tributary samples are indicated by different colors to allow them to be distinguished from one another.[306] Figure 4-3 presents the CM system used in the Study Area. Reference area data are shown on the right side of the Study Area data, separated by a vertical line. Data for each reference area are shown as a vertical stack of points rather than as a function of CM in the reference area because the primary purpose is to show the range of concentrations in each reference area compared to the Study Area.

- **Box plots** present the data grouped according to reach within the Study Area (based on individual tributaries and CM in the main stem, as discussed in Section 4.1) and the Phase 1 and Phase 2 reference areas. Their primary purpose is to show the distribution of data within a Study Area reach or a reference area and facilitate comparison among reaches between the Study Area and reference areas. The boxes span the 25th and 75th percentiles of the data (i.e., the interquartile range). The horizontal line through each box indicates the median. Whiskers extend beyond the boxes to the 10th and 90th percentiles. All individual data values beyond the 10th and 90th percentiles are presented as individual symbols.[307] The coloring scheme of the boxes matches the coloring of the longitudinal profiles. Sample counts are posted at the top of the panel. The boxes present the data from each tributary individually and the main stem for CM 0 – 1, CM 1 – 2, and CM 2+ (which includes the Turning Basin).
- **Cross plots** present a comparison of two sets of data (e.g., surface versus subsurface sediment chemical concentrations within a core or one analyte or location versus another) whose purpose is to allow for a visual comparison of the relationship between the two sets of data, including whether one is systematically higher or lower than the other. In cases where cross plots are used to evaluate systematic differences between two sets of data (e.g., to evaluate whether surface or subsurface concentrations are generally higher

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[306] In the case of surface sediment data, a variation of the longitudinal profile is used to allow for a more detailed view of spatial patterns. Each figure is presented on two pages: on the first page, the main stem and English Kills are shown, and on the second, the other tributaries are shown on separate plot panels.

[307] For sample sets with less than or equal to ten values, whiskers are not provided, and all data lying outside the interquartile range are presented as individual symbols. For sample sets with less than five values, boxes are not presented, and all data are presented as symbols.

within sediment cores), a 1:1 line is plotted to visually identify the position of each dataset relative to the other. For such cross plots, a binomial statistical test is included to test the null hypothesis that the two datasets are equally distributed above and below the 1:1 line and are not systematically different. If the significance level (p value) from the binomial test is < 0.05, the null hypothesis is rejected, indicating that one of the two datasets is systematically higher than the other. In other cases where cross plots are used and a meaningful functional relationship may be expected between two sets of data (e.g., TPCB congener versus TPCB Aroclor for samples analyzed by both methods), a regression line (typically a linear regression line, but in some cases the regression is performed in log space) and corresponding coefficient of determination (r^2) value are shown.

- **Vertical profiles** present data by depth in the sediment bed in the Study Area, with depth on the vertical axis (zero defined as the surface of the sediment bed) and the data value on the horizontal axis. These profiles may represent individual locations (e.g., a single sediment core location) or groupings of locations (e.g., all sediment cores from a given reach). For the latter case, vertical profiles are shown using box plots, in which samples are grouped into panels by reach (tributary and main stem separately). Depth intervals for the boxes are based on the generally used subsurface sampling intervals in the Phase 2 RI Work Plan Volume 1 (Anchor QEA 2014a). For each box, all core segments that include the specified depth are included (i.e., no length weighting was performed). For example, all core samples with a depth interval that includes 80 cm are grouped into a box plotted at a depth of 80 cm. These box plots allow for comparison of the central tendency and distribution of data values within each area by depth in a simple manner.

References

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